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Appendix E
Aquatic Resources

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THE POTENTIAL EFFECTS OF GLOBAL CLIMATE CHANGE ON THE UNITED STATES:

APPENDIX E - AQUATIC RESOURCES

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OFFICE OF POLICY, PLANNING AND EVALUATION
U.S. ENVIRONMENTAL PROTECTION AGENCY
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PREFACE

The ecological and economic implications of the greenhouse effect have been the subject of discussion within the scientific community for the past three decades. In recent years, members of Congress have held hearings on the greenhouse effect and have begun to examine its implications for public policy. This interest was accentuated during a series of hearings held in June 1986 by the Subcommittee on Pollution of the Senate Environment and Public Works Committee. Following the hearings, committee members sent a formal request to the EPA Administrator, asking the Agency to undertake two studies on climate change due to the greenhouse effect.

> One of the studies we are requesting should examine the potential health and environmental effects of climate change. This study should include, but not be limited to, the potential impacts on agriculture, forests, wetlands, human health, rivers, lakes, and estuaries, as well as other ecosystems and societal impacts. This study should be designed to include original analyses, to identify and fill in where important research gaps exist, and to solicit the opinions of knowledgeable people throughout the country through a process of public hearings and meetings.

To meet this request, EPA produced the report entitled *The Potential Effects of Global Climate Change on the United States*. For that report, EPA commissioned fifty-five studies by academic and government scientists on the potential effects of global climate change. Each study was reviewed by at least two peer reviewers. The Effects Report summarizes the results of all of those studies. The complete results of each study are contained in Appendices A through J.

Appendix	**Subject**
A	Water Resources
B	Sea Level Rise
C	Agriculture
D	Forests
E	Aquatic Resources
F	Air Quality
G	Health
H	Infrastructure
I	Variability
J	Policy

GOAL

The goal of the Effects Report was to try to give a sense of the possible direction of changes from a global warming as well as a sense of the magnitude. Specifically, we examined the following issues:

- sensitivities of systems to changes in climate (since we cannot predict regional climate change, we can only identify sensitivities to changes in climate factors)
- the range of effects under different warming scenarios
- regional differences among effects
- interactions among effects on a regional level

- national effects
- uncertainties
- policy implications
- research needs

The four regions chosen for the studies were California, the Great Lakes, the Southeast, and the Great Plains. Many studies focused on impacts in a single region, while others examined potential impacts on a national scale.

SCENARIOS USED FOR THE EFFECTS REPORT STUDIES

The Effects Report studies used several scenarios to examine the sensitivities of various systems to changes in climate. The scenarios used are plausible sets of circumstances although none of them should be considered to be predictions of regional climate change. The most common scenario used was the doubled CO_2 scenario ($2XCO_2$), which examined the effects of climate under a doubling of atmospheric carbon dioxide concentrations. This doubling is estimated to raise average global temperatures by 1.5 to 4.5°C by the latter half of the 21st century. Transient scenarios, which estimate how climate may change over time in response to a steady increase in greenhouse gases, were also used. In addition, analog scenarios of past warm periods, such as the 1930s, were used.

The scenarios combined average monthly climate change estimates for regional grid boxes from General Circulation Models (GCMs) with 1951-80 climate observations from sites in the respective grid boxes. GCMs are dynamic models that simulate the physical processes of the atmosphere and oceans to estimate global climate under different conditions, such as increasing concentrations of greenhouse gases (e.g., $2XCO_2$).

The scenarios and GCMs used in the studies have certain limitations. The scenarios used for the studies assume that temporal and spatial variability do not change from current conditions. The first of two major limitations related to the GCMs is their low spatial resolution. GCMs use rather large grid boxes where climate is averaged for the whole grid box, while in fact climate may be quite variable within a grid box. The second limitation is the simplified way that GCMs treat physical factors such as clouds, oceans, albedo, and land surface hydrology. Because of these limitations, GCMs often disagree with each other on estimates of regional climate change (as well as the magnitude of global changes) and should not be considered to be predictions.

To obtain a range of scenarios, EPA asked the researchers to use output from the following GCMs:

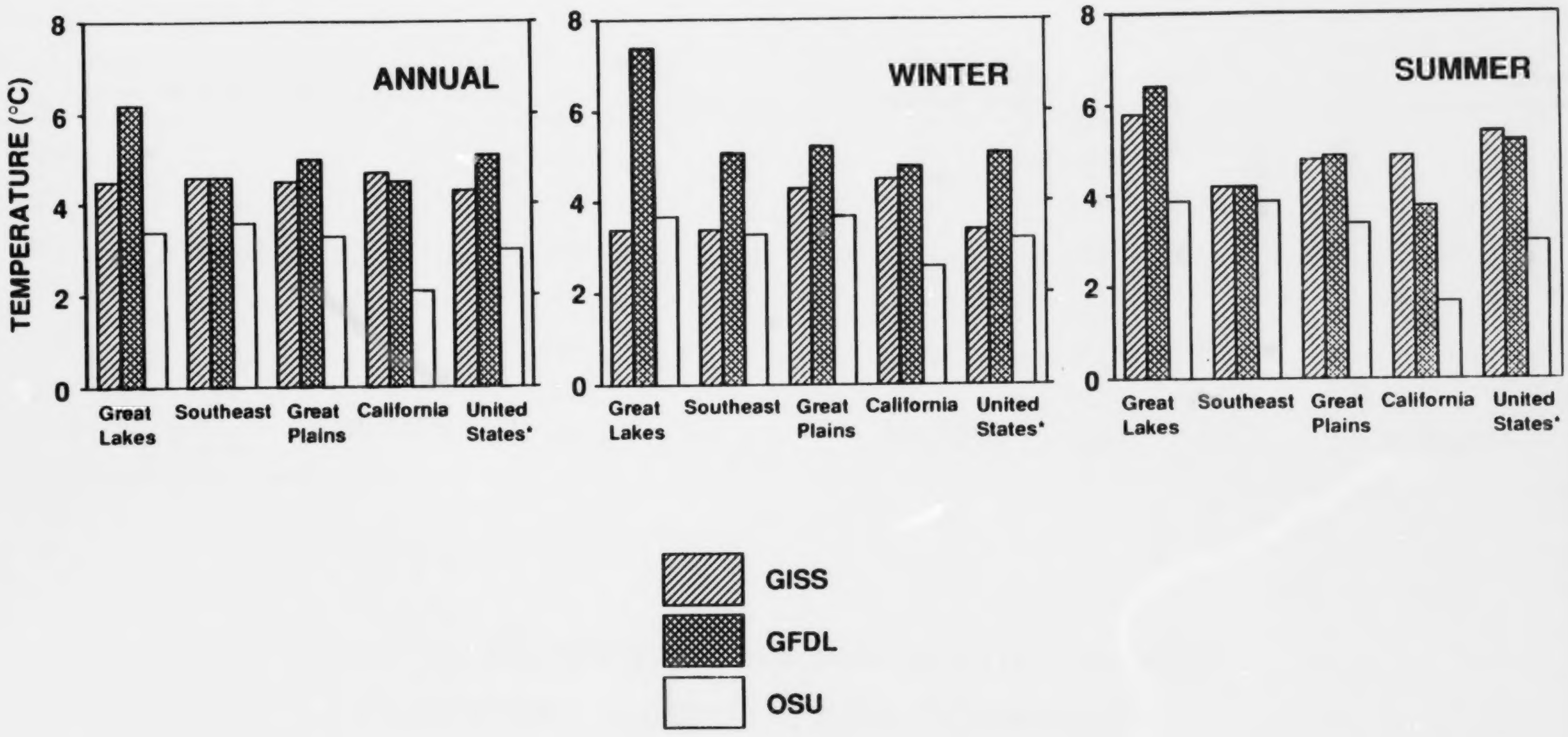
- Goddard Institute for Space Studies (GISS)
- Geophysical Fluid Dynamics Laboratory (GFDL)
- Oregon State University (OSU)

Figure 1 shows the temperature change from current climate to a climate with a doubling of CO_2 levels, as modeled by the three GCMs. The figure includes the GCM estimates for the four regions. Precipitation changes are shown in Figure 2. Note the disagreement in the GCM estimates concerning the direction of change of regional and seasonal precipitation and the agreement concerning increasing temperatures.

Two transient scenarios from the GISS model were also used, and the average decadal temperature changes are shown in Figure 3.

FIGURE 1. TEMPERATURE SCENARIOS

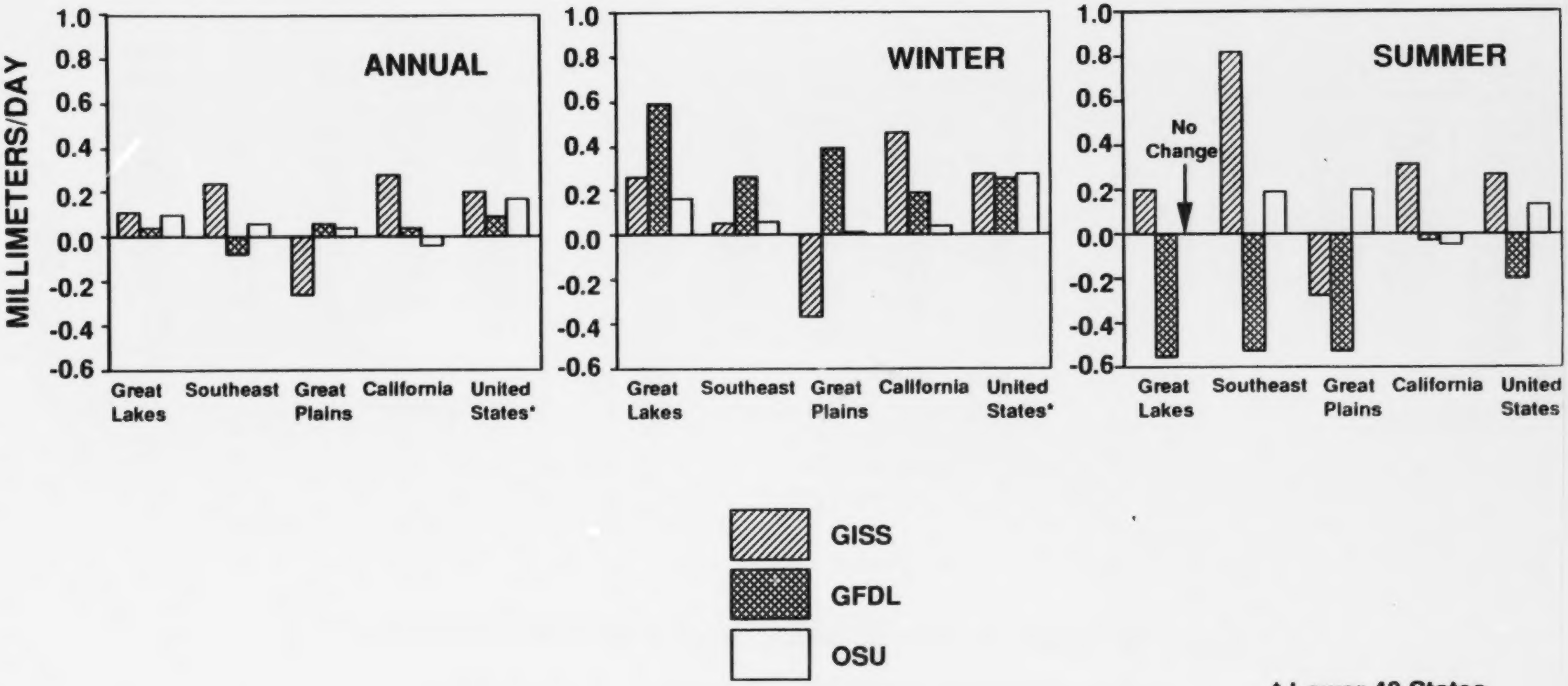
GCM Estimated Change in Temperature from $1xCO_2$ to $2xCO_2$



* Lower 48 States

FIGURE 2. PRECIPITATION SCENARIOS

GCM Estimated Change in Precipitation from $1xCO_2$ to $2xCO_2$



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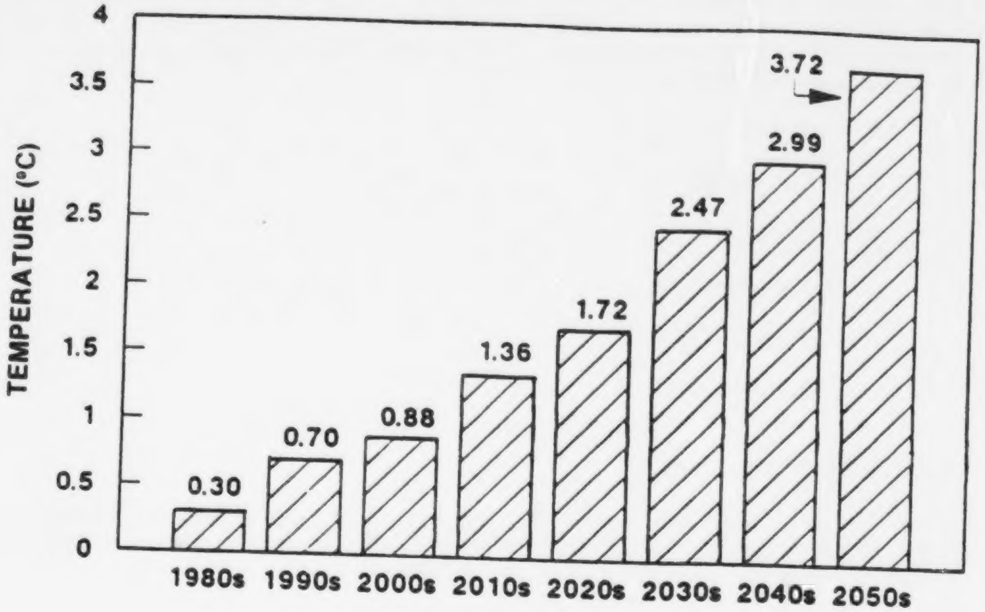
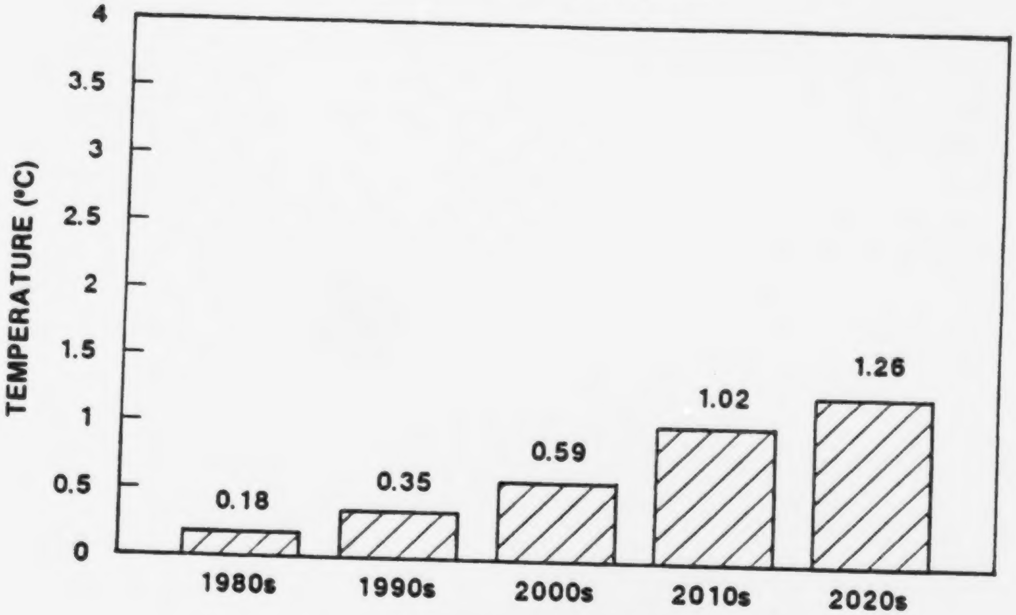


TRANSIENT SCENARIO A



TRANSIENT SCENARIO B

FIGURE 3. GISS TRANSIENTS "A" AND "B" AVERAGE TEMPERATURE CHANGE FOR LOWER 48 STATES GRID POINTS.

EPA specified that researchers were to use three doubled CO_2 scenarios, two transient scenarios, and an analog scenario in their studies. Many researchers, however, did not have sufficient time or resources to use all of the scenarios. EPA asked the researchers to run the scenarios in the following order, going as far through the list as time and resources allowed:

1. GISS doubled CO_2
2. GFDL doubled CO_2
3. GISS transient A
4. OSU doubled CO_2
5. Analog (1930 to 1939)
6. GISS transient B

ABOUT THESE APPENDICES

The studies contained in these appendices appear in the form that the researchers submitted them to EPA. These reports do not necessarily reflect the official position of the U.S. Environmental Protection Agency. Mention of trade names does not constitute an endorsement.

THE EFFECTS OF GLOBAL CLIMATE CHANGE ON THE WATER QUALITY OF MOUNTAIN LAKES AND STREAMS

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FINDINGS[1]

We researched the effects of global climate change on the water quality of subalpine lakes and streams in California. Using our long-term data bases of Castle Lake and Ward Creek, Lake Tahoe, both in the mountains of northern California, we were able to relate our water quality records to climatic variables. We then applied several scenarios of global warming to the empirical models to predict the potential effects of climatic warming on water quality in these systems.

Our results indicated that the changing precipitation patterns and global warming predicted by various climate change scenarios would have a detrimental effect on water quality for our systems. Algal growth rates (primary productivity) in Castle Lake increased under all scenario projections. Average increases ranged from 16% to 87% over control values. The enhanced algal productivity under scenario conditions resulted from the combined effects of an increase in the length of the growing season and a change in hydraulic loading. Our preliminary analysis of Ward Creek did not reveal an increase in runoff and nutrient concentrations in the scenario conditions. However, since we were not able to incorporate temperature change in our Ward Creek model, the results are much less certain than for Castle Lake.

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[1] Although the information in this report has been funded wholly or partly by the U.S. Environmental Protection Agency under Contract No. CR-814581-01-0 with the University of California, Davis, it does not necessarily reflect the Agency's views, and no official endorsement should be inferred from it.

CHAPTER 1

INTRODUCTION

PERSPECTIVE

The emphasis of our research on the ecological effects of climate change has been directed toward anticipated changes in the water quality of mountain lakes and streams. The long-term limnological research of the Lake Tahoe and Castle Lake research groups of University of California, Davis has provided a unique opportunity to examine an extensive limnological data base that spans several decades. In our previous research, we have established strong, quantifiable relationships between climatic variables and water quality, and it is logical that further research can be applied to the EPA effort in modeling the ecological effects of changing climate.

Our work interprets the long-term limnological data bases of a northern California mountain lake and stream. Our first data set, from subalpine Castle Lake, spans the longest period of time and has yielded the strongest relationships. In this analysis we have taken advantage of 29 years of limnological and precipitation data to establish relationships between annual phytoplankton productivity and climatic variables. Extensive limnological research over the years has been focused on understanding the mechanisms concerned with nutrient dynamics and system productivity (e.g., Goldman, in press; Goldman et al., submitted; Goldman and de Amezaga, 1984; Axler et al., 1981). The lake is clearly sensitive to climatic change, with most of the interannual variability in heat budget (Strub et al., 1985) and primary productivity accountable to variation in climate (Goldman et al., submitted). One of the key variables responding to climate change is the extent and timing of ice cover. Long-term records have established clear responses between ice cover and climate (Semtner, 1984; Rannie, 1983; Kuusisto, 1987). Long-term records have been used empirically to project lake ice changes, but not subsequent water quality effects in changing climate scenarios (Kuusisto, 1987)

For our analysis of mountain stream water quality, we have integrated our 12 years of data on Ward Creek, a tributary of Lake Tahoe, to yield precipitation-streamflow and streamflow-nutrient concentration relationships useful in predicting the water quality changes associated with changing climate. Ward Creek has also shown large water quality fluctuations in response to drought or high-precipitation years (Leonard et al., 1979; Byron and Goldman, 1986).

The study of the causes of variability in the water quality of lakes and streams is of both theoretical and practical interest. Most previous research has focused on the causes of long-term changes, but interannual variability is receiving increasing attention as well. For example, long-term records have revealed water quality changes in lakes due to eutrophication (Edmondson and Lehman, 1981; Goldman, 1986, 1981, 1974) and the effects of climatic fluctuations (Wetzel, 1981; Strub et al., 1985; Goldman et al., submitted). In addition, streams may experience long-term changes in water quality due to anthropogenic disturbances of watersheds (Leonard et al., 1979) and changing patterns of atmospheric deposition (Smith et al., 1987).

The realization that unprecedented global climate changes brought about by projected CO_2 and trace gas increases will occur, has spawned a new body of research concerned with potential secondary effects (e.g. Ramanathan, 1988). Hydrologic research has focused on projections of changes in water supply and subsequent societal impacts (Cohen, 1986, 1987; Kuusisto, 1987). Studies in California have projected significant hydrologic changes under doubled atmospheric CO_2 scenarios (Gleick, 1987). In this study, we propose to establish empirical models of water quality and climatic variables from our long-term data sets. These models will allow us to add water quality projections to the ongoing hydrologic and water supply modeling of other researchers.

SITE DESCRIPTIONS

Castle Lake is typical of subalpine lakes in the Western United States. It is located at an elevation of 1657 m in a protected cirque basin in the Klamath mountains of northern California. The lake has a surface area of 0.20 km^2, a maximum depth of 35 m, and a mean depth of 11.4 m. It is meso-oligotrophic, dimictic, and ice-covered in winter. Ward Valley, Lake Tahoe, is also a rather typical subalpine watershed of the central Sierra Nevada. Ward Creek runs shallow, cold, and clear for the majority of the year. The relief of the watershed ranges from 2700 m on the surrounding peaks to 1898 m at the level of its drainage into Lake Tahoe. The total area is 2510 ha, contributing approximately 6% of the annual surface runoff to the lake. The Ward Valley watershed has been glaciated, and soil parent material is primarily andesitic and basaltic volcanic rock (Leonard et al., 1979).

The remaining portion of our report includes two major sections. First, we describe the development of the empirical water quality models for Castle Lake and Ward Creek, Lake Tahoe, from our two long-term data sets. Second, we apply our empirical models to the climatic variable outputs provided by EPA to examine the effects of climate change scenarios on the water quality of the lake and stream. In each case, the results are discussed with regard to model limitations and to the environmental and socioeconomic implications of the predicted water quality changes.

CHAPTER 2

METHODS

THE EMPIRICAL EFFECTS MODELS

Castle Lake

The 29-year data record at Castle Lake has given us an excellent opportunity to understand the importance of long-term variability in the water quality of a subalpine lake. Climatic events appear to dominate limnological variation in Castle Lake, at least as indicated by the primary productivity of the phytoplankton community. Primary production makes an excellent water quality indicator since it simultaneously reflects local weather conditions, nutrient availability, and the size of the producer populations. Further, it indirectly reflects the activities of other trophic levels, such as grazing and nutrient regeneration. In addition, it is of direct importance to our investigation of oligotrophic, subalpine lakes, since primary productivity is causally related to water clarity, an important measure of water quality in these systems (Goldman, in press). Although results based on primary productivity are not immediately transferable to other lakes, in part because of the rarity of long-term productivity records, the variable gives us an immediate measure of eutrophication, perhaps the most important overall water quality indicator. Our empirical water quality model for Castle Lake gives us the ability to forecast primary production from a knowledge of climatic variation and previous production.

Our long-term record at Castle Lake results from samples collected over the entire water column at a single deep station every 5 days during the summer months and irregularly during other months since 1959. Our model uses summer primary productivity estimates for the period of June 1 to September 30 of each year (measured to be 75% of total annual production). Temperature and precipitation estimates were provided by the nearby Mt. Shasta City weather station (NWS #5983).

Model development

Annual primary production at Castle Lake is clearly related to climate. In particular, Castle Lake has exhibited anomalous (unusually high or low) primary production during ENSO (El Niño/Southern Oscillation) years. In order to investigate the dependence of annual primary production on specific climate variables, the 1960-1984 series of primary productivity measurements was first prewhitened with a Markov process to remove an autocorrelation at lag 1 year. The prewhitened series was created as follows:

$$WP_t = (P_t - a_0) - a_1 * (P_{t-1} - a_0) \qquad (1)$$

where WP_t is the series of prewhitened production at year t, P_t (mg C $m^{-2}d^{-1}$) is the series of actual primary production at year t, and the a_i are constants; $a_0 = 355 \pm 29$ and $a_1 = 0.385 \pm 0.189$.

A significant cross-correlation at lag zero was detected between prewhitened primary production and both winter snowfall and annual hydraulic load. Although snowfall and hydraulic load are associated, the relation between prewhitened primary production and each of these variables is still significant, even when adjusted for the other variable. Snowfall is important because of its effect on ice breakup time; hydraulic load because of its effect on outwash rates in the spring. The autocorrelation in primary production may arise from persistence of certain phytoplankton populations from one year to the next (Vincent and Goldman, 1980).

Based on these associations with the prewhitened series, a time series regression model was built relating measured annual primary production to snowfall and precipitation. Primary production was transformed with logarithms to stabilize variance; the autocorrelation was addressed by including lag 1 primary production; climate effects were represented by the product of January-April snowfall and January-May total precipitation; and a moving-average model for the residuals was included. All coefficients in the model were significantly different from zero at the $p < 0.05$ level. The initial model accounted for 81% of the variability in the logarithm of primary production:

$$(1-\delta * B) \ln P_t = w_0 + w_1 * S_t * H_t + n_t \qquad (2)$$

where B is the backshift operator (i.e. $B x_t = x_{t-1}$), P_t (mg C $m^{-2}d^{-1}$) is primary production during year t, S_t is January-April snowfall (actual snow depth in cm, not water equivalent) during year t, H_t (cm) is January-May hydraulic loading during year t, n_t is the residual series, and δ, w_0, and w_1 are constants. P_t is measured summer productivity (June-September). In 1968, when measurements were made year-round, these summer months accounted for 75% of the annual production.

Since some autocorrelation was found in the residual series at lag 2, we selected a model for the residuals as:

$$n_t = a_t + \theta * a_{t-2} \qquad (3)$$

where a_t is a series of independent white noise $N(0, \sigma_a)$ and θ is constant. The overall initial model using inputs of measured productivity, snowfall, and total precipitation became:

$$(1-\delta * B) \ln P_t = w_0 + w_1 * S_t * H_t + (1 + \theta * B^2) a_t \qquad (4)$$

Snowfall Estimates

Output from the global climate models does not include snowfall, which is required for the final Castle Lake production model (Eq. 4). We therefore attempted to develop a forecasting model for monthly snowfall, based on monthly values for the climate model outputs of total precipitation and temperature. Using the actual monthly data for the years 1960-1984, the same years for which Eq. 1 was developed, a variety of approaches was tried, including time-series regression models for individual months, and regression models developed using data for all winter months. A second-degree polynomial model applicable for all winter months was chosen; it performed almost as well as the more complicated time-series regression models developed for individual months:

$$S = b_1 * H + b_2 * H^2 + b_3 * H * T \qquad (5)$$

where S is monthly snowfall (cm snow depth), H is monthly total precipitation (cm), and T is mean monthly temperature (°K). The constants are $b_1 = 191 \pm 27$, $b_2 = -0.100 \pm 0.015$, and $b_3 = -0.672 \pm 0.097$. The variance accounted for (r^2) is 56%, and the residuals exhibit no significant autocorrelation or other internal structure.

Model Using Estimated Snowfall

An attempt was made to develop a new forecasting model for annual production that excluded snowfall and used temperature (monthly maximum, minimum, and average). However, no improvement could be made over the form of Eq. 4, even when S_t was estimated by summing values determined from Eq. 5 for January through

April. As a result, the parameters in Eq. 4 were estimated again using these predicted values of S. The resulting model explained much less of the variance ($r^2 = 58\%$), but the parameters remained significantly nonzero ($p < 0.01$) and a moving average term was not needed to remove residual structure (i.e., $\theta = 0$). The new parameter values were $\delta = 0.571a \pm 0.142$, $w_0 = 2.85 \pm 0.81$, and $w_1 = -(3.78 \pm 0.79) \cdot 10^{-5}$. The structure of the model is given in Table 1.

Ward Creek, Lake Tahoe

Our data records for precipitation and streamflow for Ward Valley encompass 12 years of daily streamflow and nutrient load estimates and storm-event based precipitation sampling. In developing the water quality models, we limited the record to our longest unbroken string of data: 1980-1986. The period of modeling is further limited by the weather records for this study, which end in 1984. Approximately 100-150 stream samples were taken each year to generate gauge height-concentration curves that allowed us to use continuously recorded gauge height to estimate average daily concentration values. Precipitation data were provided by the U.S. Weather Bureau, Tahoe City (NWS #8758). While this is not a particularly long data set compared to our lake data, it has nevertheless yielded significant regressions relating water quality to climatic variables.

We had to examine several hydrologic parameters in order to establish the water quality regressions. Our basic two-part approach was first to establish a relationship between precipitation and streamflow and then to describe the relationship between flow and nutrient concentrations. Our data record is long enough to include a drought year (1981) as well as very high water years (1982, 1983). Thus we feel that the significant relationships we have established encompass most of the present range of climatic variation.

The precipitation-flow model for Ward Creek used precipitation at lag 3 and 6 months and a moving average model for the residuals (1 month) to predict monthly total streamflow (Figure 1, Table 2). The model was well behaved, i.e., all coefficients were significant and the residuals had all structure removed. However, only 55% of the variation was accounted for, and it was not possible to incorporate temperature into the model in a statistically acceptable manner. Neither present month nor lagged temperature terms added to the explanation of variance. The 3- and 6-month lag terms for precipitation were necessary to incorporate snowmelt accumulation and melt effects into the model (and, thus, incorporate some potential temperature effects).

The second step of our model estimates various aspects of annual average stream nutrient concentrations as predicted from stream runoff. In this case we were not limited by the weather data, and our period of analysis was from January 1980 through September 1986 (81 observations). Stream discharge was significantly associated with both soluble phosphorus and nitrate concentrations for this period (Figures 2, 3; Table 3). As might be expected, stream discharge was also significantly associated with loading of both soluble phosphorus and nitrate (Figures 4, 5; Table 4). The increase in predictive capability with loadings is due to the fact that load is the product of stream discharge and nutrient concentration.

Limitations of the Empirical Models

In the case of Castle Lake, the model is limited to the prediction of annual total values of algal production. Seasonal effects are lost. Still, the model was created empirically, using enough years of data to validate the relationships for a wide range of climatic variation. It offers a good unbiased estimate of direction and overall magnitude of changes when using scenario data. For Ward Creek, the seasonal effects are preserved by the use of monthly data. However, the relative lack of change in cumulative precipitation in the climate change scenarios and inability to incorporate temperature in the models precluded the use of the monthly data in any attempts to model water quality changes.

The models make use of effects of precipitation and temperature change but do not incorporate the direct effects of a doubling of CO_2 in the atmosphere into their predictions. Atmospheric CO_2 increase is only potentially important in the case of the Castle Lake primary productivity model, but even in that case the effects of carbon limitation on algal growth are unlikely to be significant. Subalpine lakes like Castle are relatively rich

Table 1. Castle Lake Primary Production Model

PPR = "Annual" primary production (June - September)
LPPR = Ln(PPR)

$TMPOi_t$ = mean temperature (^0K) for year t and month i (i = 1,...,4)

$PRCOi_t$ = total precipitation (cm) for year t and month i (i = 1,....5)

$SNWOi_t = MAX(191*PRCOi_t-0.0996*PRCOi_t{}^2-0.672*PRCOi_t*TMPOi_t, 0)$

$PRC_t = PRCO1_t+PRCO2_t+PRCO3_t+PRCO4_t+PRCO5_t$

$SNW_t = SNWO1_t+SNWO2_t+SNWO3_t+SNWO4_t$

$LPPR_t = MAX(2.85+0.571*LPPR_{t-1}-3.78*10^{-5}*SNW_t*PRC_t, 0)$

$PPR_t = EXP(LPPR_t)$

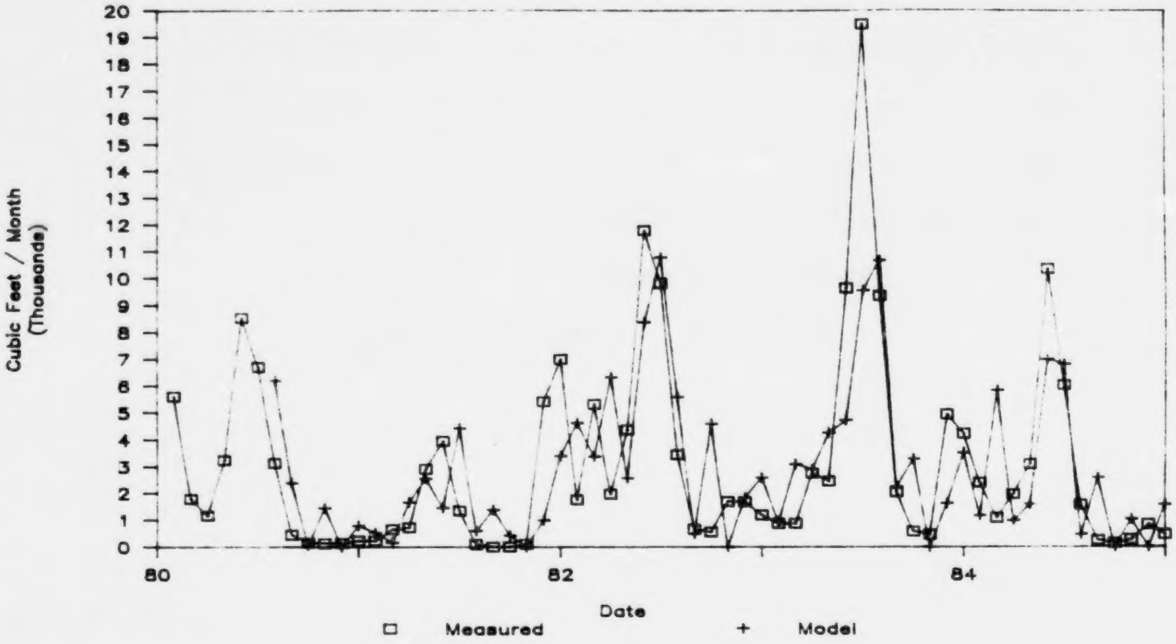

Figure 1. Prediction of water discharge from precipitation, Ward Creek, Lake Tahoe.

Table 2. Ward Creek Streamflow Prediction

Dependent Variable is Water Discharge

1980.07 - 1984.12

Number of observations: 54

Convergence achieved after 5 iterations

VARIABLE	COEFFICIENT	STD. ERROR	T-STAT.	2-TAIL SIG.
Precipitation (-3)	403	85.1	4.73	0.000
Precipitation (-6)	376	75.3	5.00	0.000
Residuals:				
Moving Average (1)	0.725	0.140	5.17	0.000
R-squared	0.545			
S.E. of regression	2600			
Durbin-Watson stat	1.98			
F-statistic	30.6			

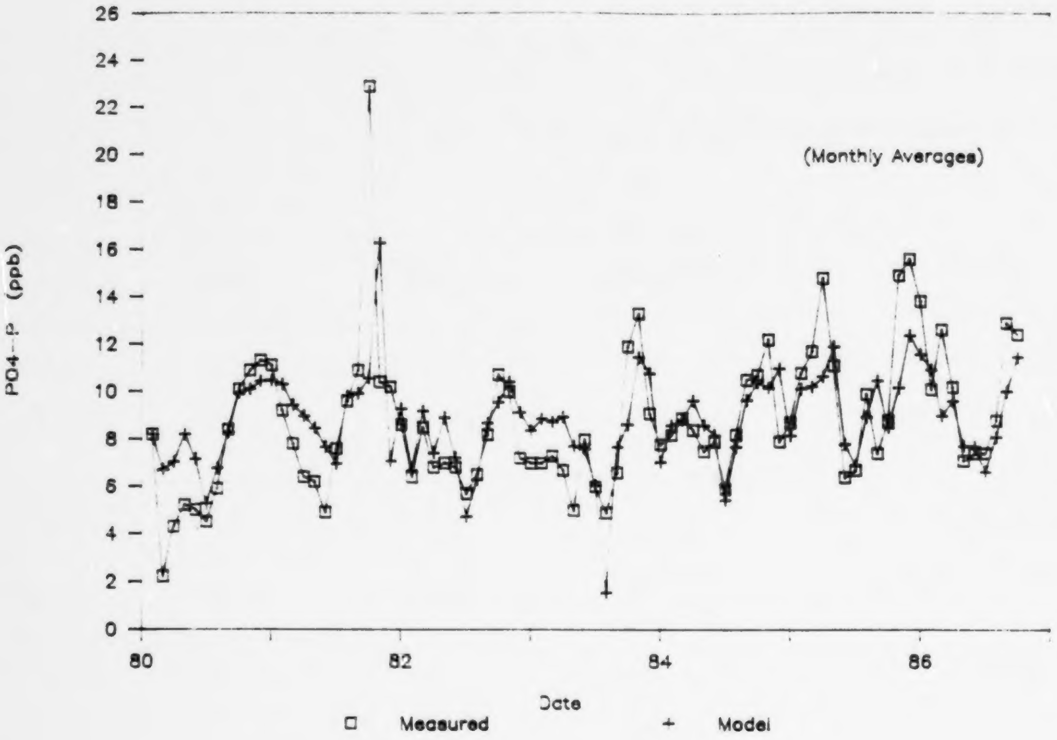

Figure 2. Prediction of soluble phosphorus concentrations from water discharge, Ward Creek, Lake Tahoe.

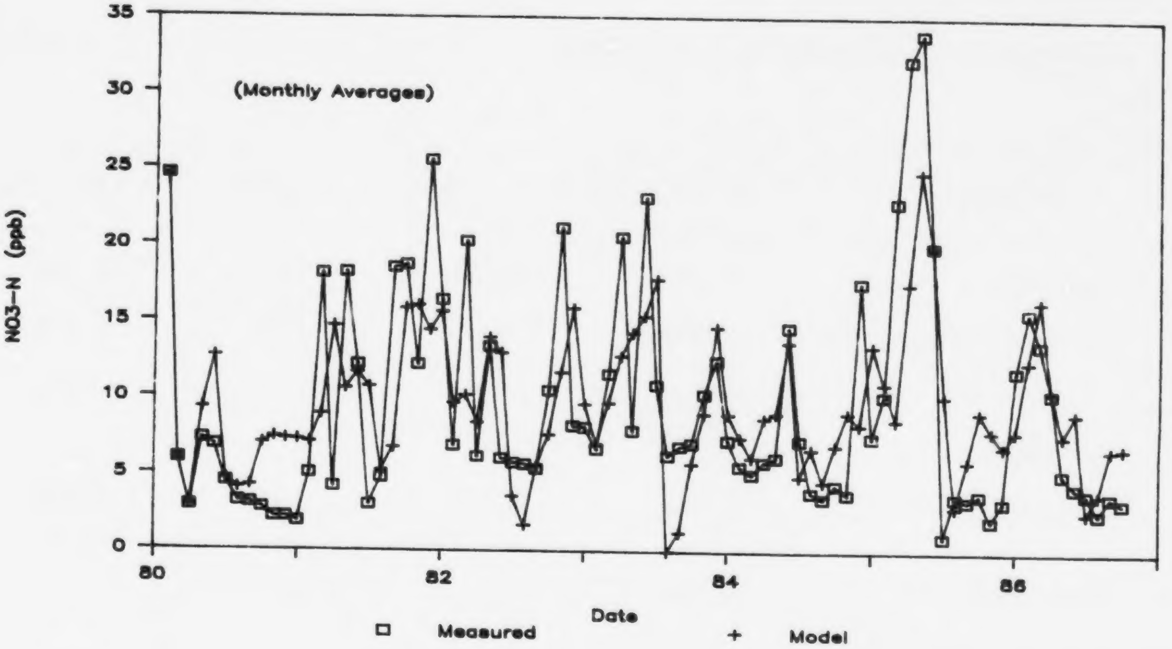

Figure 3. Prediction of nitrate concentrations from water discharge, Ward Creek, Lake Tahoe.

Table 3a. Ward Creek Soluble Phosphorus Concentration Prediction

```
Dependent Variable is Soluble Phosphorus Concentration

1980.02 - 1986.09

Number of observations:  80

Convergence achieved after 3 iterations

-----------------------------------------------------------------------------
   VARIABLE              COEFFICIENT    STD. ERROR      T-STAT.    2-TAIL SIG.
-----------------------------------------------------------------------------
Constant                     10.1          0.350          28.8         0.000

Water Discharge (-1)     -0.000436       7.81D-05        -5.59         0.000
- - - - - - - - - - - - - - - - - - - - - - - - - - - - - - - - - - - - - - -
Residuals:

  Moving Average (1)       0.503          0.114          4.41         0.000
-----------------------------------------------------------------------------
R-squared                  0.422

S.E. of regression         2.39

Durbin-Watson stat         1.95

F-statistic                28.1
-----------------------------------------------------------------------------
```

Table 3b. Ward Creek Nitrate Concentration Prediction

Dependent Variable is Nitrate Concentration

1980.04 - 1986.09

Number of observations: 78

Convergence achieved after 5 iterations

VARIABLE	COEFFICIENT	STD. ERROR	T-STAT.	2-TAIL SIG.
Constant	11.2	1.07	10.5	0.000
Water Discharge	0.000782	0.000236	3.31	0.002
Discharge (-1)	-0.000939	0.000235	-4.00	0.000
Discharge (-3)	-0.000461	0.000193	-2.39	0.020
Residuals:				
Moving Average (1)	0.462	0.119	3.90	0.000
Moving Average (2)	0.298	0.118	2.52	0.014
R-squared	0.410			
S.E. of regression	5.73			
Durbin-Watson stat	2.00			
F-statistic	10.0			

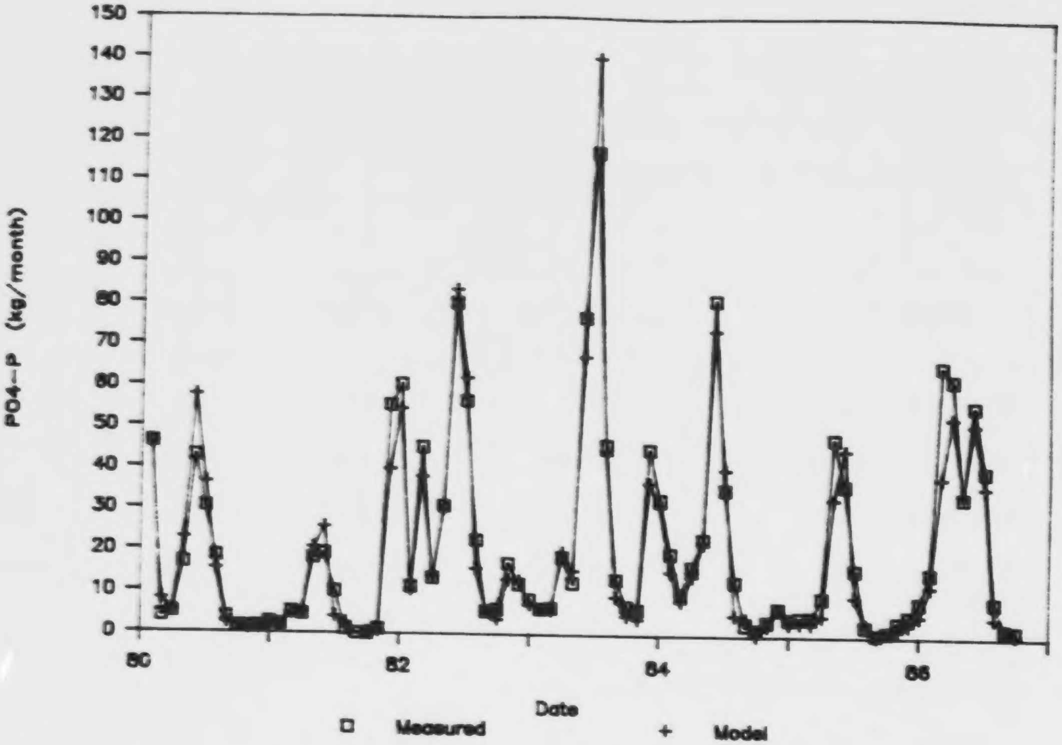

Figure 4. Prediction of soluble phosphorus loading from water discharge, Ward Creek, Lake Tahoe.

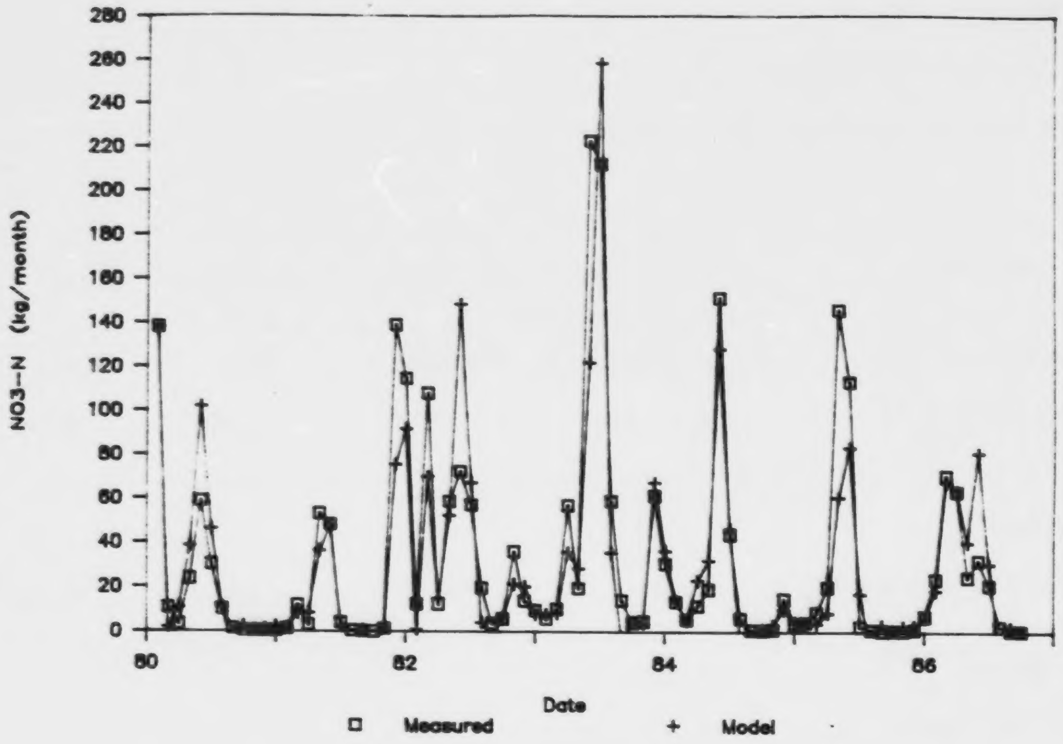

Figure 5. Prediction of nitrate loading from water discharge, Ward Creek, Lake Tahoe.

Table 4a. Ward Creek Soluble Phosphorus Loading Prediction

Dependent Variable is Soluble Phosphorus Loading

1980.02 - 1986.09

Number of observations: 80

Convergence achieved after 3 iterations

VARIABLE	COEFFICIENT	STD. ERROR	T-STAT.	2-TAIL SIG.
Water Discharge	0.00733	0.000244	30.0	0.000
Discharge (-1)	-0.000717	0.000239	-2.99	0.004
Residuals:				
Moving Average (1)	0.455	0.117	3.88	0.000
R-squared	0.933			
S.E. of regression	6.23			
Durbin-Watson stat	1.88			
F-statistic	539			

Table 4b. Ward Creek Nitrate Loading Prediction

```
Dependent Variable is Nitrate Loading
1980.02 - 1986.09
Number of observations:  80
Convergence achieved after 3 iterations
```

VARIABLE	COEFFICIENT	STD. ERROR	T-STAT.	2-TAIL SIG.
Water Discharge	0.014	0.000918	15.3	0.000
Discharge (-1)	-0.00426	0.000898	-4.75	0.000
Residuals:				
Moving Average (1)	0.264	0.118	2.25	0.028
R-squared	0.758			
S.E. of regression	23.3			
Durbin-Watson stat	1.96			
F-statistic	121			

in free CO_2 and low in bicarbonate and carbonates, and experience algal production limited by nutrient availability, not carbon (Wetzel, 1983). Increased carbon in the system could be dealt with in a more complex model. The effects of increased carbonic acid in precipitation are not incorporated into our models either. However, precipitation acidity is presently dominated by the effects of strong acids. Presumably, future increases in nitric and sulfuric acid (not estimated as part of the scenarios) would be more important in causing future ecological changes.

The Castle Lake algal productivity predictions that we have produced using the scenario results are only realistic when productivity estimates fall within or close to the historic range. As production estimates rise, the limiting factors on total production at some point will undoubtedly shift from factors associated with hydraulic washout and the length of the growing season (as the present model was developed) to nutrient input, speed of recycling, or other limitations. Nutrient inputs, for example, may change dramatically with forest vegetation and runoff changes under a doubled CO_2 regime. The present model cannot account for the possibility of changing limitations to growth associated with changing climate. Microbial processes in the soil and sediment and watershed plant communities may be expected to respond to climatic change and deserve consideration in terms of their effect on nutrient flux. A series of predictive ecological models should be incorporated into an integrative ecosystem level model in order to most accurately predict water quality changes.

THE CLIMATE CHANGE SCENARIOS

The Scenarios Used

We used the GFDL, GISS, and OSU control and $2xCO_2$, and the GISS transient A scenarios to apply to our empirical water quality models. Surface temperature and total precipitation were used from each. Although this does not represent the complete list of possible scenarios, it gives a strong consensus in direction and magnitude suggested by most of the climate change models. For Castle Lake models we used weather station data from Mt. Shasta City, California. For Ward Creek, we used Tahoe City, California, data.

The projected temperature and precipitation changes resulting from the scenarios are shown in Figures 6 through 10. In all cases, ratios of change from present conditions for each month were multiplied by historical weather data to yield scenario climatic conditions. Note that temperature in the $2xCO_2$ scenarios is higher than historical values (Figure 6), whereas precipitation changes are much less discernible (Figures 7, 8). The lack of change in precipitation can also be easily seen for the GISS transient A scenario (Figure 9). For our empirical model of Castle Lake, it is of greater significance that snowfall estimates show a strong negative trend for the transient scenario (Figure 10).

It was necessary to change the application of the scenarios for Castle Lake. As described above, our empirical model requires that productivity estimates begin in a year for which the previous summer's productivity was known. In our case, the model output was restricted to beginning in 1960, the second year of Castle Lake data collection. Thus, the measured Castle Lake productivity for 1957 was used as input to both the control and $2xCO_2$ conditions. Since the years 1951-1959 were removed from the series, we included the four additional years for which we had data (1981-1984) at the end of the data series. Thus, the series for Castle Lake runs from 1960 through 1984 for control and $2xCO_2$ data. The GISS transient A starts by using the measured 1980 Castle Lake productivity to begin the model run in 1970 for the series through 2060.

The Ward Creek models were run using the series 1980-1984 to establish precipitation and temperature conditions because our empirical models were developed for that period. The scenarios were not used to predict water quality changes for the 1951-1980 period, since the only scenario to show changing precipitation did not cause a consequent change in stream water discharge (see results for GISS $2xCO_2$ below), and temperature was not incorporated into the model. There is, therefore, no basis for examining predictions of Ward Creek water quality based on the scenario outputs. The modeling of average concentrations of stream solutes as predicted from precipitation would require more detailed information on rain, snow, and snowmelt than was available as climate model output. Instead, we present our empirical models of nutrient concentrations and load over time

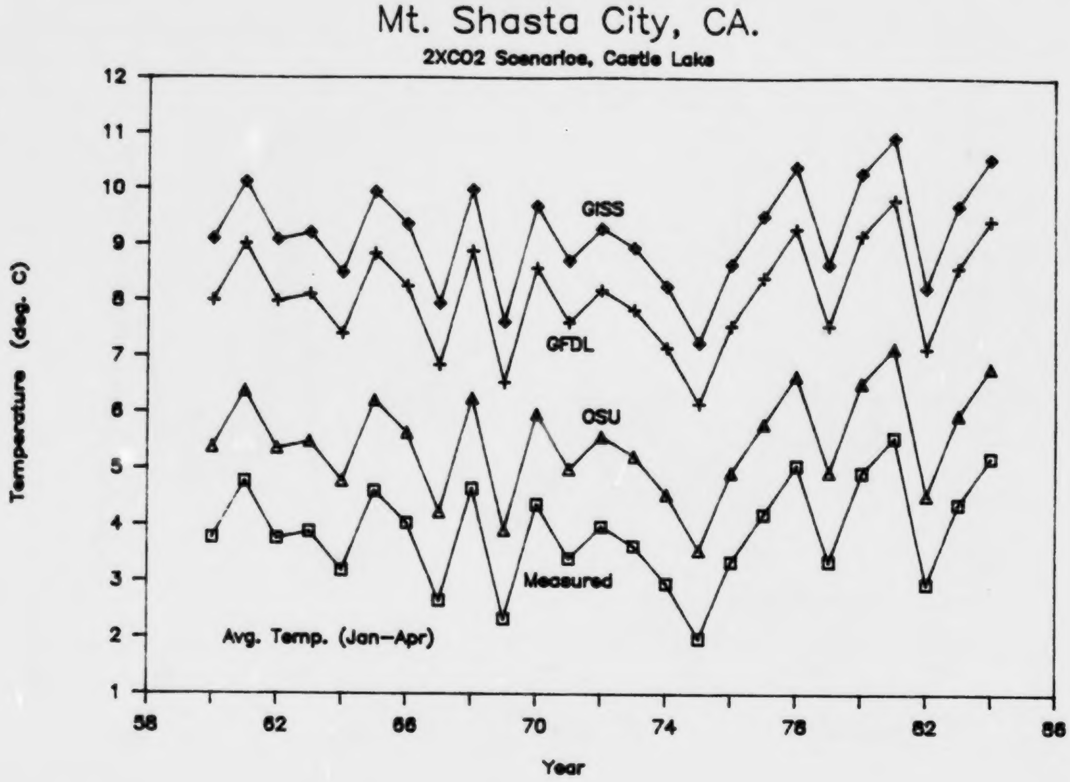

Figure 6. Measured and $2xCO_2$ scenario, January-April average temperatures, Castle Lake.

GISS gridpoint: 120W, 39.13N;
GFDL gridpoint: 120W, 40N;
OSU gridpoint: 120W, 40N.

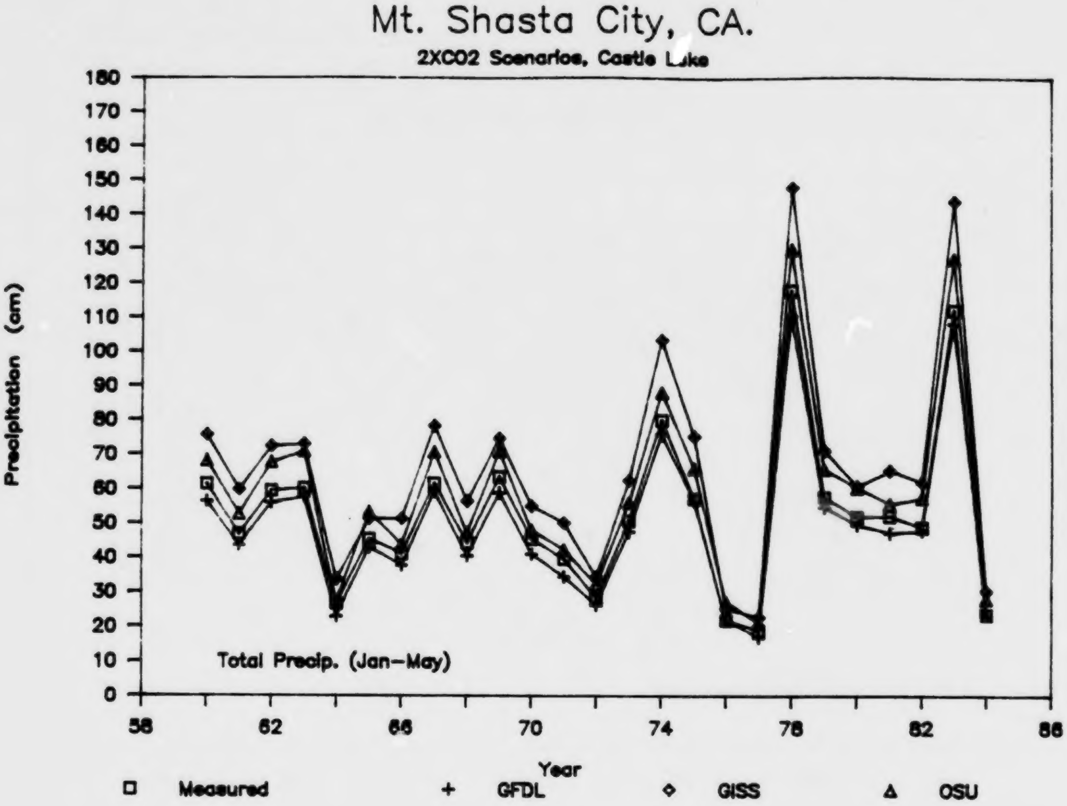

Figure 7. Measured and $2xCO_2$ scenario, January-May precipitation, Castle Lake.

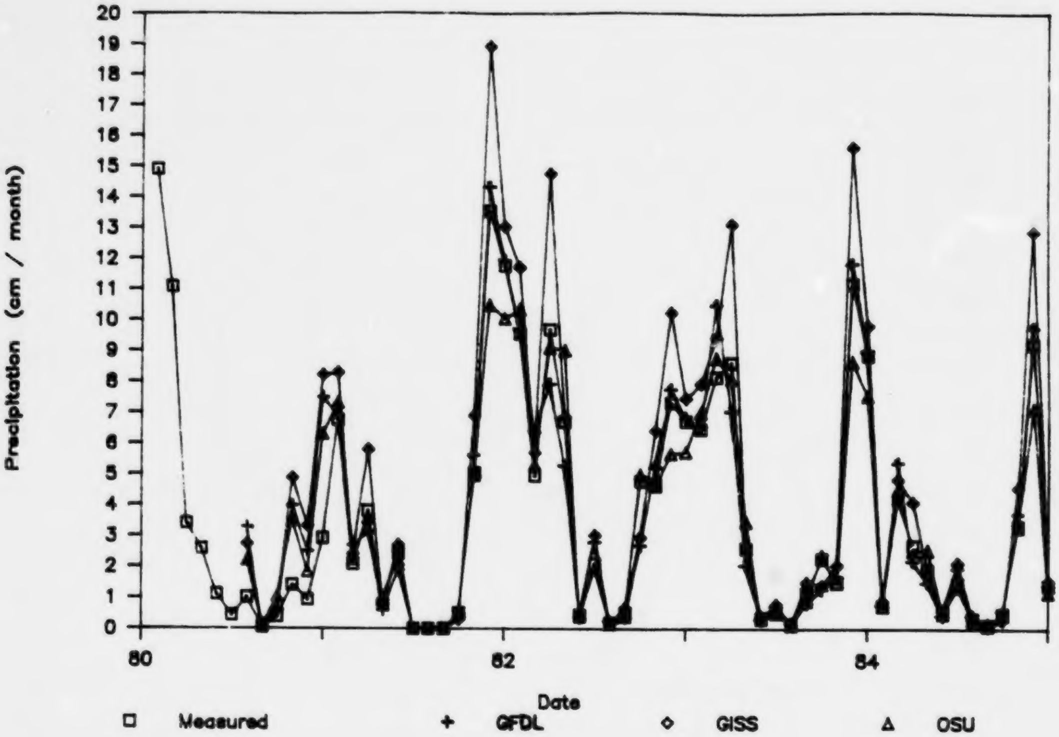

Figure 8. Measured and $2xCO_2$ scenario, monthly precipitation, Ward Creek, Lake Tahoe.

GFDL gridpoint: 120W, 35.55N;
GISS gridpoint: 120W, 39.13N;
OSU gridpoint: 120W, 36N.

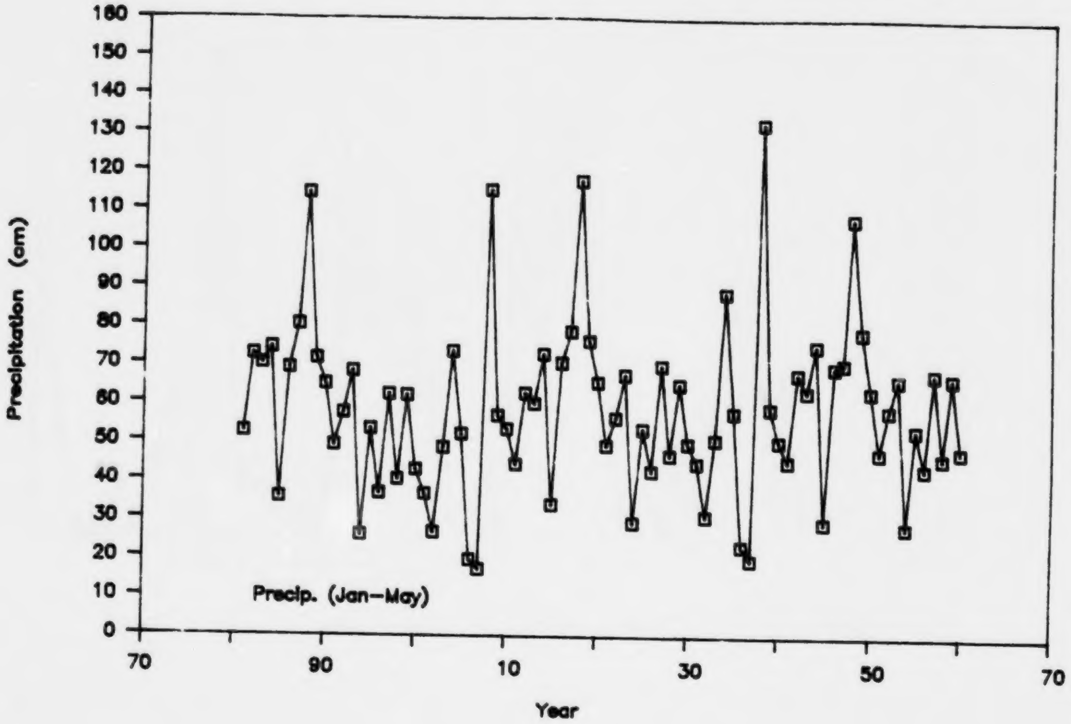

Figure 9. GISS transient A, monthly precipitation, Castle Lake.

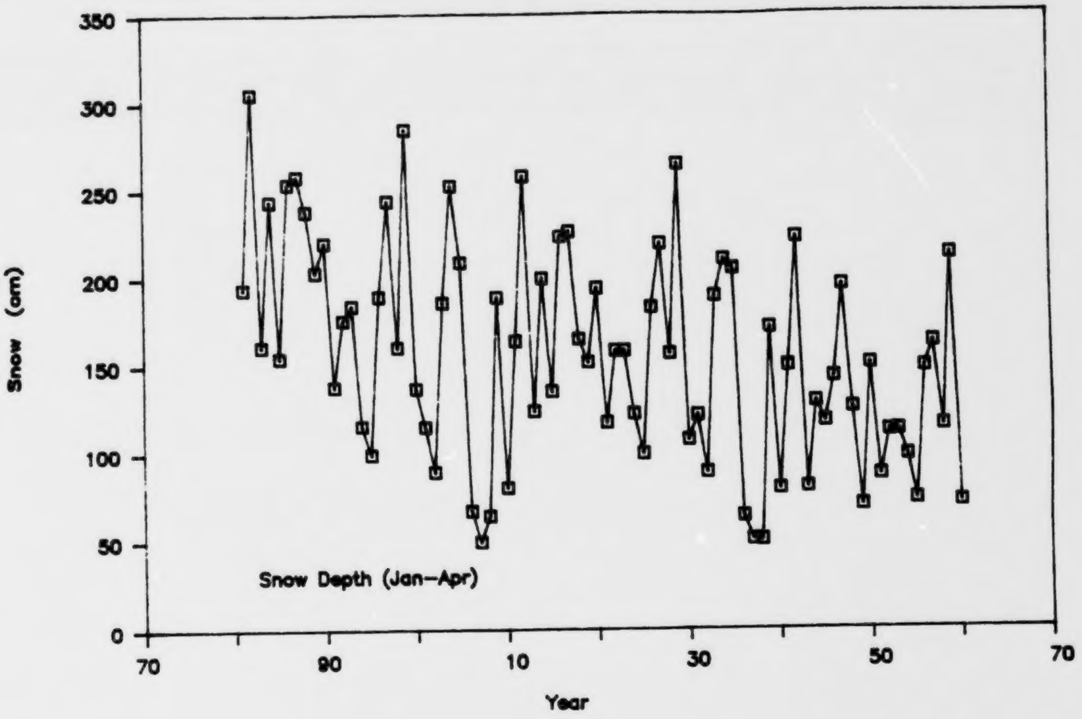

Figure 10. Monthly snow estimates calculated using precipitation and temperature results from GISS transient A, Castle Lake.

for the years 1980-1984 as predicted from the relationships between monthly stream water discharge and nutrient concentrations.

Issues Resulting From the Scenarios

Our empirical models require a knowledge of snow accumulation and melt to adequately predict water quality parameters. The present models would be greatly aided by refinements in the scenario outputs that would allow more accurate predictions of precipitation, temperature, and snowfall on a fine-grained scale. The coarse nature of the large global climate model grids yields ratios of change in parameters that are of dubious value in predicting change in mountainous subregions of the larger grids.

The years 1960-1984 are adequate for establishing a full range of baseline conditions. In terms of hydrology and water quality parameters, they encompass an extensive range of drought, "normal," and high-water years. Although these represent a deviation from the basic 1951-1980 data set, they are still representative of the same hydrologic range.

CHAPTER 3

RESULTS

CASTLE LAKE

Annual primary production was forecast for five cases using the equations shown in Table 1: (1) actual conditions (Table 5, Figure 11); (2) climate output from the the three $2xCO_2$ models (Table 6, Figure 12); and (3) climate output from the GISS transient scenario A (Figure 13). Table 1 contains the actual equations used in the simulation. Note that for all simulations, snowfall was set to 0 when Eq. 2 predicted negative snowfall. The results for $2xCO_2$ are summarized in Figure 14.

The simulation reproduced the actual primary production values quite accurately (Figure 11), although the model had difficulty recreating the large swings experienced during the period 1969-1977, during which time four major ENSO events occurred (1969, 1972, 1976, 1977) (Strub et al., 1985). It should be noted that Figure 11 represents a dynamic simulation that propagates errors in estimated primary production through the lag 1-year term.

The three $2xCO_2$ simulations behaved in a similar fashion, although quite variable in magnitude (Figure 12). The increase in temperature caused by the $2xCO_2$ conditions resulted in an increase in mean annual primary production of 69.0% (GFDL), 87.0% (GISS), and 16.3% (OSU) over estimated "control" conditions for the $2xCO_2$ scenarios. These changes are close to our measured historical range, but show a consistent direction of change toward higher production under conditions of higher atmospheric CO_2 concentrations. It is possible to substitute an average $2xCO_2$ value for our measured 1959 value to start the $2xCO_2$ runs. The mean $2xCO_2$ primary production values increase slightly under those conditions, from 0.3% (OSU) to 3% (GISS) greater than the averages shown in Table 6.

The GISS transient A scenario produced an increasing but variable primary productivity estimate for the period 1970-2059 for Castle Lake, showing the expected strong 30-year scale of variation inherent in the 1951-1980 data base (Figure 13). In all scenarios, therefore, we project increasing productivity. Whereas the increases may be small at Castle Lake, the variability between the lakes is unknown. The direction of change is clear. The implications of eutrophication are discussed below.

WARD CREEK, LAKE TAHOE

Since our empirical models for stream water quality did not use temperature, we were limited in our ability to apply our results to the $2xCO_2$ and transient scenarios. Of the climate conditions we examined for the Tahoe City weather station, the GISS $2xCO_2$ precipitation estimates stood apart as representing consistently wetter conditions. We tested whether the GISS-altered precipitation regime would affect stream discharge by using the empirical regression equation for precipitation-discharge described above.

The results show isolated periods of deviation of GISS discharge from measured streamflow but no consistent trends in magnitude or direction (Figure 15). Apparently, the slightly elevated precipitation regime of the GISS scenario is not enough to significantly alter runoff conditions in our models, perhaps reflecting the relatively small watersheds of mountain lakes. If we could incorporate temperature as well as precipitation into our precipitation-runoff model, there would be a greater likelihood of demonstrating some effect of the scenario conditions. With our present models we have no further basis for examining water quality parameters under the scenario conditions.

Table 5. Modeled vs. Measured Primary Production, Castle Lake

1960 - 1984

Number of observations: 25

SERIES	MEAN	S.D.	MAXIMUM	MINIMUM
Measured Production	349	99.6	569	136
Modeled Production	340	68.9	468	216

	Covariance	Correlation
PPR,PPR1XO	427	0.648

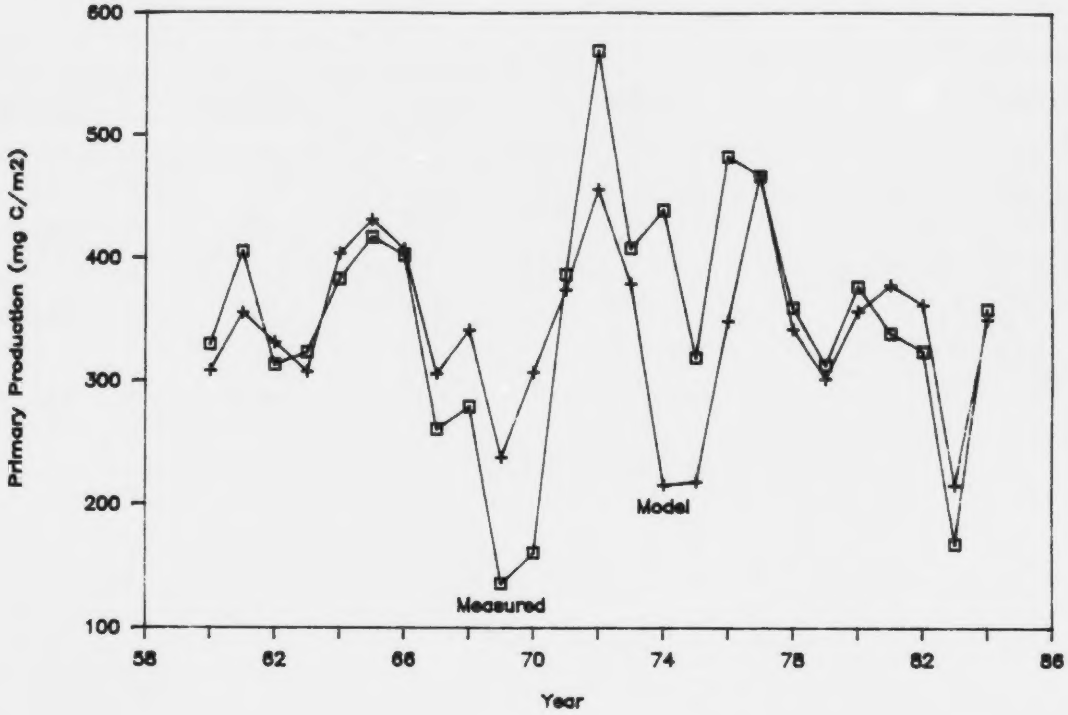

Figure 11. Measured and modeled annual primary production, Castle Lake, 1960-1984.

Table 6. 2xCO$_2$ Scenario Results, Castle Lake: Production Estimates

1960 - 1984

Number of observations: 25

SERIES	MEAN	S.D.	MAXIMUM	MINIMUM
GFDL	576	65.5	677	422
GISS	636	59.0	720	446
OSU	396	72.7	493	250

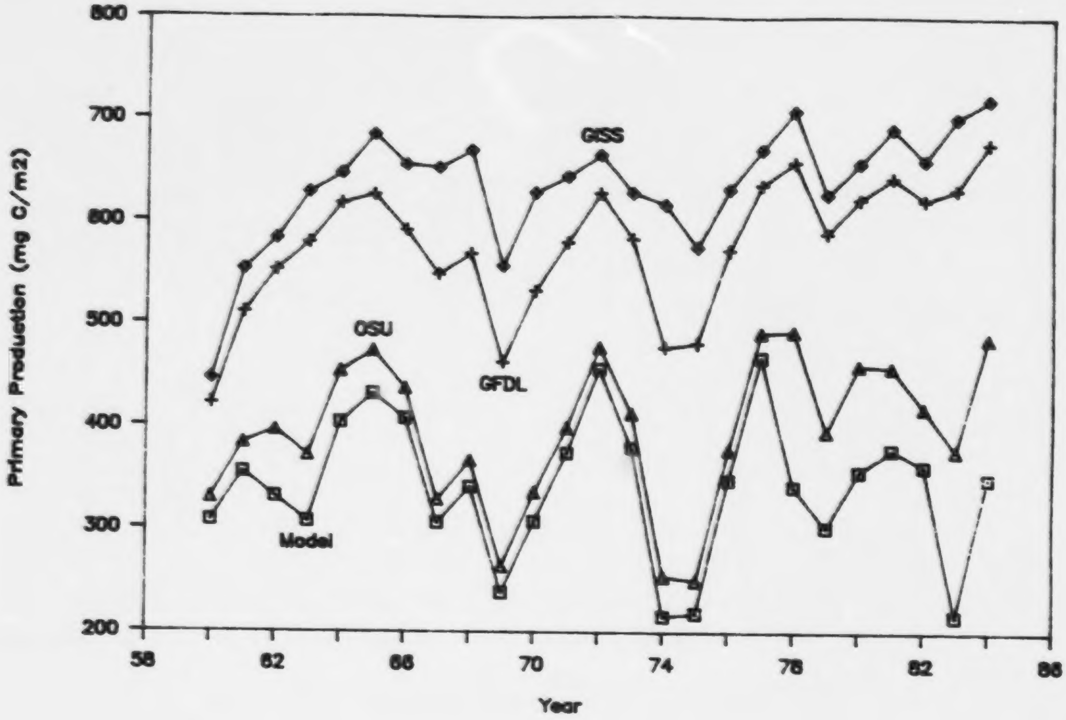

Figure 12. Control and $2xCO_2$ scenario, annual primary production estimates, Castle Lake.

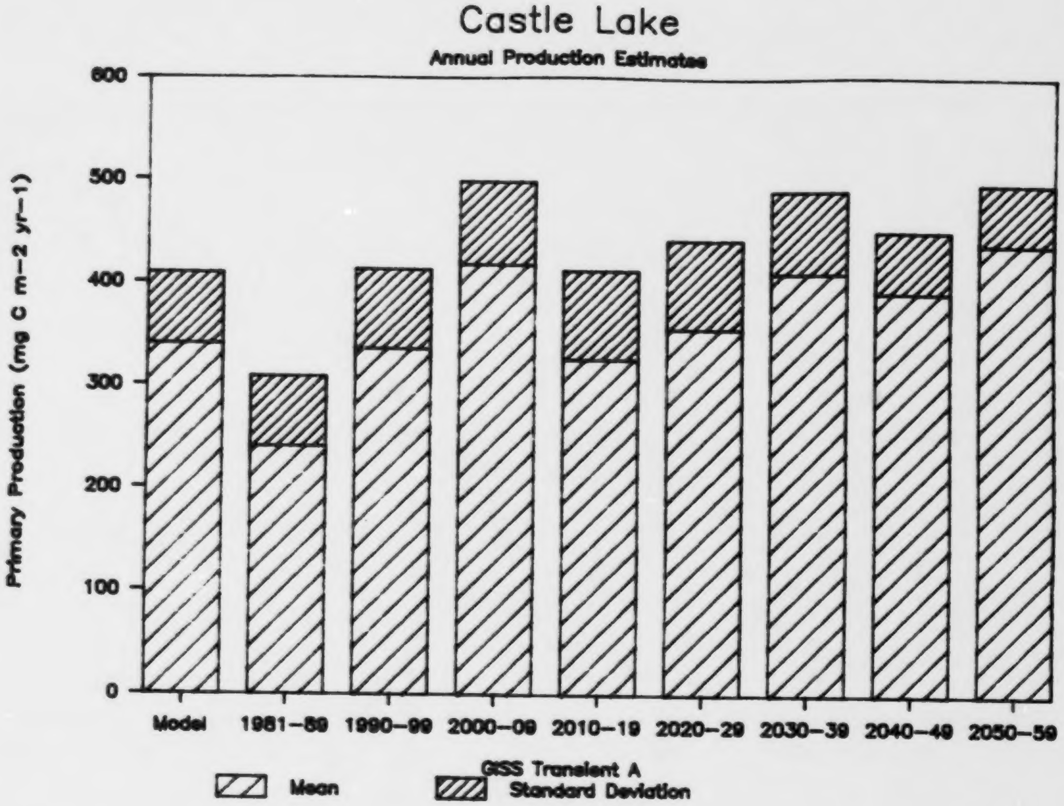

Figure 13. GISS transient A, annual primary production estimates, Castle Lake, plus model "control" values. Transient values are decadal means. In all cases, one standard deviation is shown as added to the mean.

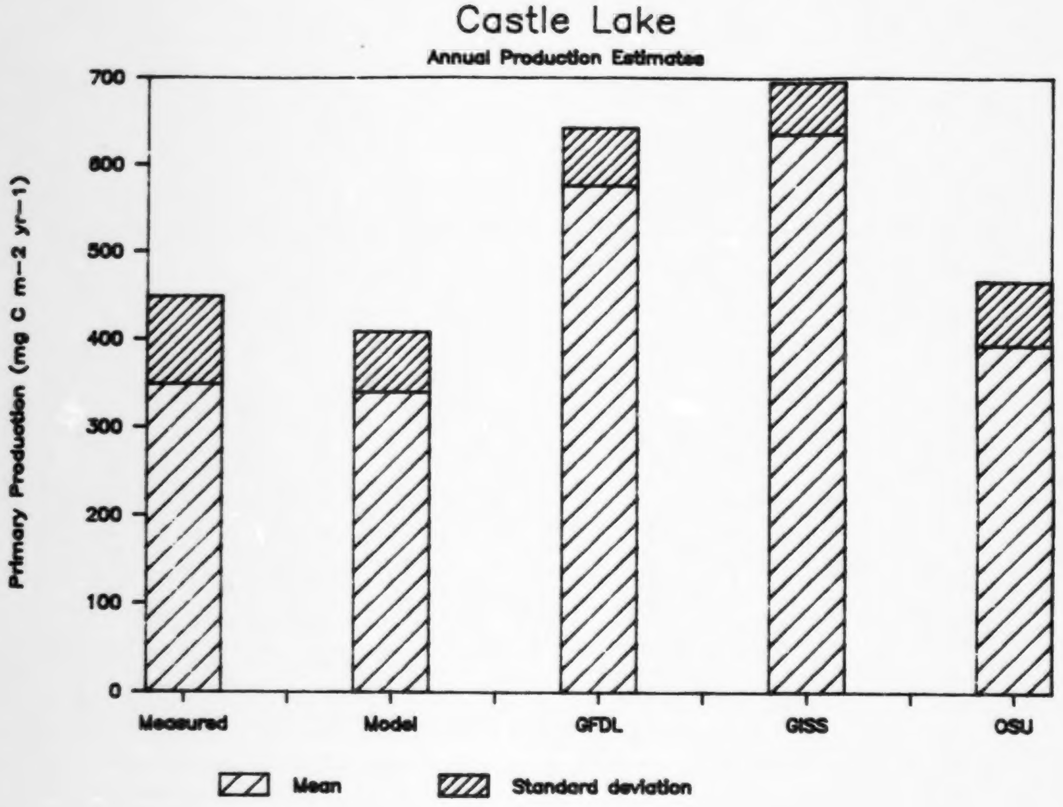

Figure 14. Summary of production estimates: all scenarios plus measured and model "control" values. One Standard deviation is shown as added to the mean.

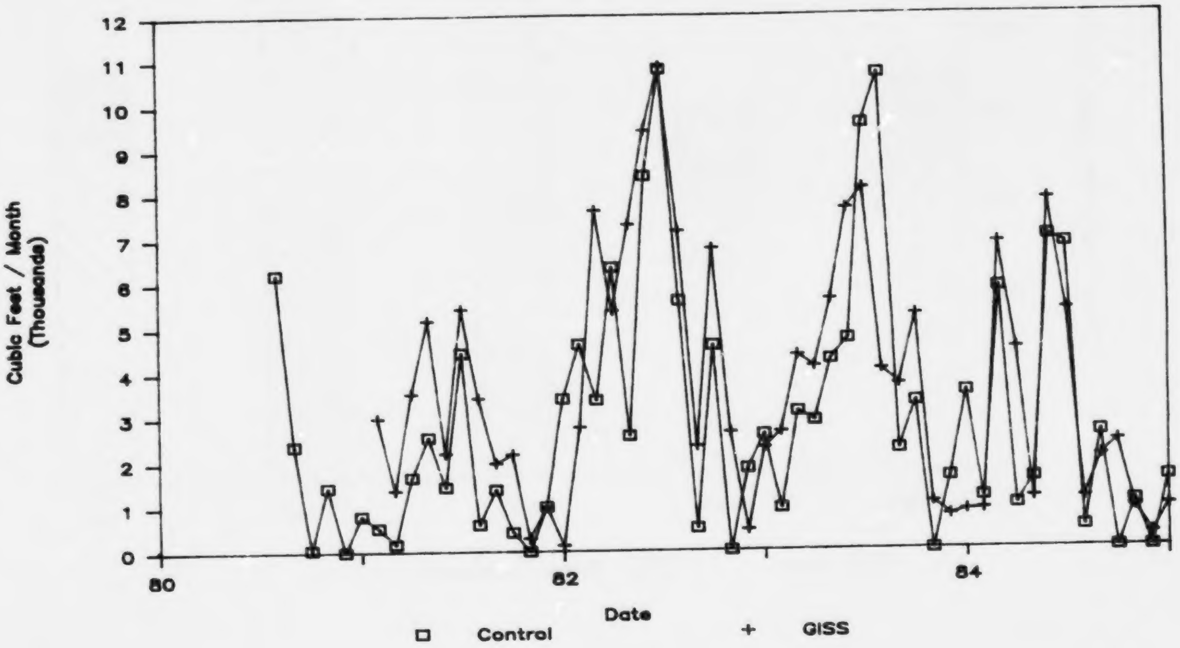

Figure 15. Modeled "control" and GISS $2xCO_2$ water discharge estimates, Ward Creek, Lake Tahoe.

CHAPTER 4

INTERPRETATION

The changes in Castle Lake associated with the climatic change scenarios were driven primarily by temperature. The increased temperature under the $2xCO_2$ scenario conditions contributed to relatively less snow accumulation and consequently an early melting of the lake ice. The trend in snowfall can clearly be seen in the transient results (Figure 10). Early melting and a longer growing season resulted in higher annual primary production estimates. The relative lack of change of precipitation as compared to temperature in the scenarios lessened the impact of total precipitation on the production estimates.

The $2xCO_2$ conditions produced clearly altered productivity estimates (Figure 12) and the transient provided a definite trend (Figure 13) for Castle Lake. Short-term changes are less clear in the transient, however, owing to the masking effect of strong interannual variation. As can be seen from the summary figure, the first two 30-year periods of the transient do not produce mean production estimates more than one standard deviation from our measured historical values (Figure 14). Short-term trends occurring at the rate of the GISS transient A would be difficult to differentiate from interannual variability at a scale of 50 years or less.

In the case of Ward Creek, Lake Tahoe, water quality changes associated with climate change projections were undetectable in our analysis.

CHAPTER 5

IMPLICATIONS OF RESULTS

ENVIRONMENTAL

The implications of the projected direction of change in water quality center on the effects of enhanced primary productivity on oligotrophic mountain lakes. Our results suggest that algal production in Castle Lake would be stimulated through increasing hydraulic inputs and a longer growing season. The work on Ward Creek, although inconclusive, indicates that enhanced precipitation (as demonstrated in the GISS scenario) combined with a changing rain-snow balance (unknown in our model) could easily increase the in-stream concentrations of particulates and soluble nutrients. There is thus the possibility of increased watershed nutrient contributions as well as the direct enhancement of production through an increase in length of the growing season. It seems certain that the projected global changes associated with the increasing atmospheric CO_2 conditions would result in enhanced primary production for Castle Lake and possibly for all similar high mountain lakes.

Enhanced primary production (eutrophication) would cascade to all elements of the food web and possibly lake and downstream water quality through changing nutrient dynamics, storage, and release. As eutrophication proceeds, the dynamics of algal and bacterial blooms begin to dominate the changes in water quality. Zooplankton, macroinvertebrates, and fish all increase in biomass but are subject to large seasonal changes. Fish in the Great Lakes have been projected to change dramatically in response to climate change (Meisner et al., 1987). During stratification, decaying biota deplete oxygen in deep waters and increase the solubility of nutrients and potentially toxic compounds (e.g., metals, H_2S) which could be added to the surface water and downstream releases during mixing events. Although increased system productivity might affect such aspects of community dynamics as species diversity or the size, age structure, and dynamics of populations, the relationships are complex and require additional empirical and mechanistic models before reasonable predictions can be made.

SOCIOECONOMIC

The socioeconomic implications of mountain lake eutrophication and altered water quality are in terms of recreational uses, fisheries, and the quality of water supplies. All are adversely affected by increasing algal productivity with the exception of fisheries, which, at least in these unproductive lakes, might be enhanced by a greater system productivity. The enhancement of system productivity could result in major shifts in fish community structure as well. Lakes which are so warm and productive as to be marginal for trout production might exceed the threshold of oxygen and temperature requirements and revert to warm water fisheries conditions, as has been projected for the Great Lakes area (Meisner et al., 1987). As eutrophication proceeds, deep lakes will show oxygen depletion in the hypolimnion and lose deep fish habitat during the summer stagnation and winter ice cover. The eutrophication of mountain lakes and streams could affect local and downstream water supplies and recreation in a multiplicative fashion. Oxygen depletion, nutrient enrichment, and algal blooms may contribute to locally decreased water quality for domestic or industrial use. Subsequently, the eutrophication of upstream areas will have a negative impact on downstream receiving lakes and streams independent of climate-induced changes. The negative impact on water supplies will also be immediately evident in recreational use. Although limited eutrophication could enhance fisheries, the deterioration of water quality will negatively affect other recreational uses.

CHAPTER 6

POLICY IMPLICATIONS

Lake water quality and eutrophication are important considerations in the regulation and planning of recreation areas. Cold water trout fisheries are important for recreation and for the economics of western mountainous areas of the U.S. and Canada. A general warming of streams and lakes could eliminate or reduce spawning success of resident and anadromous salmonids at lower elevations and allow for invasion of warm water-tolerant species. Water quality degradation could necessitate policy changes regarding in-streamflows and the extent and type of water use.

CHAPTER 7

FUTURE RESEARCH

Several areas of applied research are suggested by the results of these initial studies. We have touched upon some of the potentially valuable resources of the long-term Tahoe and Castle Lake data bases, but much more can still be learned concerning the effects of climate change on mountain lake and stream water quality.

In the case of Castle Lake, we are now in the process of inputting and validating extensive data sets of zoo- and phytoplankton abundance estimates and the results of stratification on water chemistry in both lakes. Both plankton and water chemistry are indicators of water quality that are expected to show important relationships to climate. Investigation of these with predictive models should further elucidate the effects of climate change. An interpretation of this data should allow us to better understand the mechanisms by which climate influences production in lakes and streams.

In the case of Ward Creek, we have established in-stream models of water quality that only require better climate model outputs to be useful in modeling potential effects of climate change. By working with hydrologists to refine the snow accumulation and melt predictions of the climate models we should be able to provide good projections of nutrient and sediment loads under different climatic scenarios.

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POTENTIAL RESPONSES OF GREAT LAKES FISHES AND THEIR HABITAT TO GLOBAL CLIMATE WARMING

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FINDINGS[1]

Global climate warming will directly influence fishes in the Great Lakes by significantly warming their aquatic habitat. These effects act on those fishes which do best in coldwater (e.g., salmon, trout and whitefish), in coolwater (e.g., yellow perch and walleye), and in warmwater (e.g., largemouth and smallmouth bass).

Three types of direct effects of altered thermal structure were examined for fishes: 1) changes in the size of their optimum thermal habitat, 2) changes in growth and prey consumption, and 3) changes in reproductive success and population size. Generally, all three thermal groups of fishes experience an increase in the size of favorable thermal habitat integrated over the annual cycle and grow at faster rates and eat more, provided they can seek out favorable temperatures and prey is available. These positive effects are apparent in all three (OSU, GISS, and GFDL) global climate model scenarios and in cold regions (northern Lake Superior), cool regions (southern Lake Michigan), and warm regions (Lake Erie) of the Great Lakes. Most of these positive effects result from an increased length of growing season. However, in the warmest part of the summer the size of favorable thermal habitat decreases for coldwater fishes like trout both in the lakes and in headwater brook trout streams. During summer, species interactions like competition and predation should be intense and may have negative effects on fishes. The increased demand for forage may also intensify species interactions and alter food web structure and efficiency.

Smallmouth bass were used as an example of potential changes in reproductive success and population size. This warmwater species, which lives in protected Great Lakes bays would experience marked increases in reproductive success and population size in cold and cool regions and would not be harmed in warm regions. Effects of climate change on the recruitment of other taxa were not evaluated.

The increase in thermal habitat for warmwater fishes may allow an increase in colonization and dispersal of exotic fishes with either negative or beneficial effects.

To estimate whether there would be a general increase in primary and secondary production to meet the increased forage requirements of the fishes, a simple ecosystem level model was applied to primary production (algae), secondary production (zooplankton), and fishery production (fish yields). In all regions of the Great Lakes primary production, zooplankton biomass, and fishery yields would be expected to increase significantly.

Thus, for most regions and most fishes, habitat and productivity would increase following climate warming. Even though productivity is expected to increase, surprises are expected involving (1) the loss of important stocks and (2) intensified species interactions due to new colonizing species and habitat constriction among the existing fish communities. The higher productivity and the potential for more permanent thermal stratification increase the possibility of anoxia in some basins and bays, excluding such regions from fishes.

Policy issues include the continued management of cold, cool, and warmwater fisheries for what appears to be higher recreational and commercial fishery opportunities, continuing to protect the water quality of the lakes especially in regard to nutrients contributing to eutrophication and toxics that accumulate in fishes, the continuing value of lake trout for rehabilitation of populations, intensified predator pressure on the forage base and intensified species interactions, and the potential problem of negative species interactions between present warm and coolwater fishes and exotic warmwater fishes that will find expanded habitats, especially in inshore areas.

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[1] Although the information in this report has been funded wholly or in part by the U.S. Environmental Protection Agency under Contract No. CR 814644-01-0 under the Office of Policy, Planning and Evaluation, it does not necessarily reflect the Agency's views, and no official endorsement should be inferred from it.

CHAPTER 1

INTRODUCTION

DESCRIPTION OF ECOLOGICAL SYSTEM

The Great Lakes lie within a relatively small watershed basin (Figure 1) that has long been the industrial heartland of North America and is now home to some 37 million people. In a world in which fresh water is becoming a scarce resource, our wealth is reflected in the fact that these lakes contain about one-sixth of all the surface fresh water of the globe, not including the glaciers and groundwaters. This chain of lakes drains through the St. Lawrence River into the Gulf of St. Lawrence, where the freshwater outflow plays a role in the reproduction and recruitment of the large herring fisheries of the gulf.

By 1950, the southern third of the basin had become a vast ecological slum because of careless agricultural, forestry, industrial, and urban development in preceding decades. A series of studies then led to binational conventions and agreements to rehabilitate the fisheries and remediate the bad practices that degraded water quality. Many billions of dollars have been spent in these efforts by the various governments at different levels. Some programs have been relatively successful: control of the predaceous sea lamprey, correction of overfishing, reduction in loadings of old-fashioned industrial wastes and of plant nutrients from sewage, and discontinuation of the use of persistent pesticides. Prospects are improving that loadings of hazardous contaminants and acidic substances will progressively be contained and reduced.

By 1950 the ecological quality of the lakes, streams, and shores was generally offensive to humans, especially in cities and towns. People had turned their backs to the lakes. Some four decades later, many people are rediscovering these ecosystems as they are now recovering from past abuses. Shore properties are escalating in value and are being redeveloped with sensitive uses in mind. Commercial fisheries have rebounded, but their value has been dwarfed by new recreational fisheries that are based, in part, on a new "manmade" association of fish species that is dominated by Pacific salmons. Parks, marinas, beaches, promenades, and condominiums along the shorelines are increasing in number and quality as the demand for them grows.

Commitment to a healthy Great Lakes ecosystem as reflected in an abundance of valued fish is part of a regional policy to redevelop, on a sustainable basis, the cities, industries, farmlands, and forests of the basin, as key to the resurgence of prosperity in the basin. In the face of this optimistic prospect, concern is growing about the likely effects of climate change.

Will the warming trigger relapses in the ecological recovery of degraded parts of the lakes? Will increased temperatures foster undesirable ecological productivity and thus impede recovery from cultural eutrophication? Will the higher temperatures create difficulties, in southern waters of the basin, for the coldwater salmons and lake trout? Will higher temperatures facilitate invasion of new species with some of them interacting to the detriment of some valued species of the present association? Will valued new species appear in the warmer habitats? Will sea lamprey benefit in Lake Superior to the detriment of lake trout and other valued fish species? Such questions come to mind as we consider the possible ecosystem impacts of climate change.

In its pristine state, the fish association of the Great Lakes could be separated into five major types:

i) The deeper, colder waters of the lakes and rivers were dominated by the salmon family (lake trout, lake whitefish, lake herring, chubs), the cod family (burbot), and the cottid family (sculpins).

ii) Waters of intermediate depth that were cool even in midsummer were dominated by the perch family (yellow perch, walleye, sauger, etc.) and the pike family (northern pike, muskellunge, etc.). The sturgeon family (lake sturgeon) and the ictalurids (bullheads, channel catfish) frequented the bottoms of cool waters.

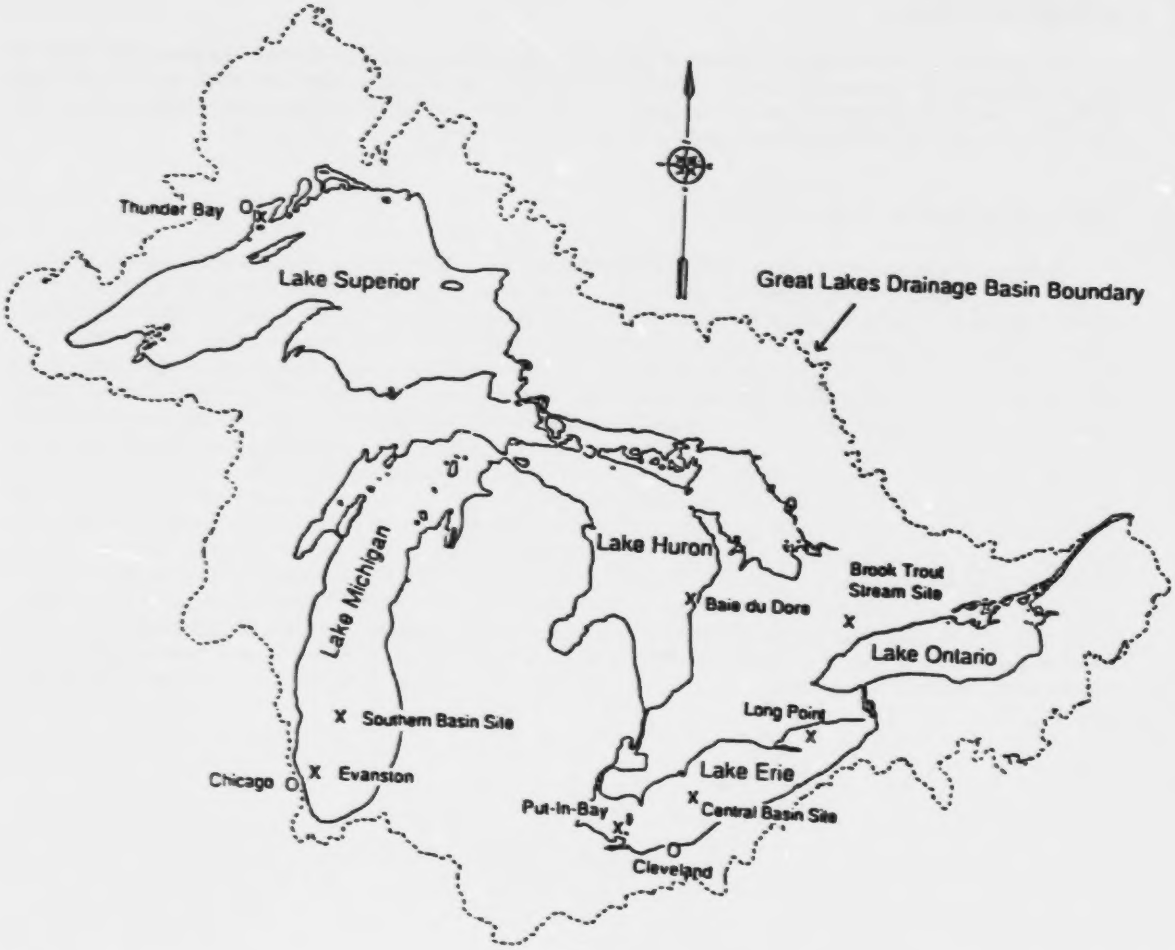
Thunder Bay
Lake Superior
Great Lakes Drainage Basin Boundary
Lake Michigan
Lake Huron
Baie du Dore
Brook Trout Stream Site
Lake Ontario
Long Point
Lake Erie
Southern Basin Site
Evanston
Chicago
Put-In-Bay
Central Basin Site
Cleveland

Figure 1. Great Lakes drainage basin and sites referred to in text. Symbols: 0 = air temperature data site, X = water temperature data site.

iii) In a few of the warmest waters of moderate depth, the "percichthyid" family (white bass) fluctuated in abundance over the decades.

iv) Shallow, warm inshore waters, especially of bays and streams, were the home of the sunfish family (sunfishes, crappies, smallmouth bass, largemouth bass, etc.) and the cyprinid family (minnows).

v) In the forested coldwater tributary streams, again the salmon family (brook trout) and cottid family (sculpins) predominated.

At the outset, it seems likely that each of these five types of associations will react somewhat differently to climate warming in a particular locale. Coldwater habitats may shrink toward the north in the lakes and upstream into the headwaters; coolwater habitats may be shifted northward and upstream; warmwater habitats may expand from the south northward and from the protected bays into the lake proper.

ORGANIZATION OF THIS REPORT

Ecological phenomena in our report are addressed in a hierarchic context across several levels of organization, progressively from the organism level, to the population level, to the ecosystem level. First, the report considers the potential influence of altered thermal structure on the size of the habitat within the thermal niche of fishes which prefer and do best in cold, cool, or warm waters. Second, our report considers how the body growth and the prey consumption of coldwater, coolwater, and warmwater fishes may be changed by living near-shore in Great Lakes neritic habitats altered by climate change scenarios. Third, for a warmwater fish that lives in the shallowest inshore habitats and for which the best models were available, our report evaluates the response of a population of fishes in respect to reproductive success and population size. Fourth, the report steps back from the species level and estimates the response of phytoplankton production, zooplankton biomass, and fishery yields as entire trophic levels to climate warming scenarios. To seek generality among global climate models, three climate change scenarios were applied: Oregon State University's (OSU) general circulation model, the general circulation model from the Geophysical Fluid Dynamics Laboratory (GFDL), and the general circulation model from Goddard Institute of Space Studies (GISS). To seek generality across the Great Lakes, simulations were made for a warm region in Lake Erie, a cool region in southern Lake Michigan or Lake Huron, and a cold region in northern Lake Superior. Finally, the report discusses what we do not know or have not addressed especially in regard to the possibility of surprises that could occur in the dynamic food webs perturbed by altered fish species interactions. A recent review paper by Meisner et al. (1987) is appended for general reference.

CHAPTER 2

THERMAL NICHE

GREAT LAKES

Fishes of the Great Lakes have been grouped into three broad thermal guilds for which a central temperature has been estimated by Hokanson (1977): coldwater ca 15.0°C; coolwater ca 24.0°C; and warmwater ca 28.0°C. Body temperatures of fishes are determined by the temperatures of the water in which they live, and changes in their environmental temperatures directly influence their survival, physiology, and behavior (Fry, 1971; Hokanson, 1977; Magnuson et al., 1979). Fish species within a thermal guild have preferred temperature ranges in which growth is maximal. The median temperature of the preferred range of a species is where the species will be found at greatest densities if food resources are not limiting and interspecific competition is not high. The thermal niche of a species was defined as ± 2.0°C of the median temperature by Magnuson et al. (1979).

The amount of thermal habitat within a fish's thermal niche has been shown to be a statistically significant determinant of fishing yield (Christie and Regier, 1988). Estimates of maximum sustained commercial fish yield of lake trout, lake whitefish, and walleye were directly related to the amount of lake bottom area or pelagic volume contained within the optimal temperature ranges of these species in a set of 21 large North American lakes.

Altered thermal structure of the Great Lakes owing to climate warming could increase or decrease the size of optimum thermal habitats available for important fishes. This would ultimately change commercial and sport fishing yields of valued species. The above empirical model of thermal niche is used here to estimate the amount of thermal habitat available for the thermal guilds of Great Lakes fishes before and after applying the global climate warming scenarios. Expected changes in potential yield of several commercial fishes are estimated from the Christie and Regier (1988) scaling of maximum sustained yields to summer thermal habitat.

Methods

Potential effects of climatic warming on the size of thermal habitats of fishes representing several of the thermal guilds were simulated using thermal structure data from southern Lake Michigan (McCormick 1988) and the central basin of Lake Erie (Blumberg, 1988). These results were subsequently used to estimate changes in fishery yields.

Thermal habitat was estimated for individual species from three thermal guilds. The thermal niche was defined in two alternative ways, one narrow and the other broad: (1) the optimum ±2°C, which includes the temperatures within which a fish in a preference tank in the laboratory will spend 2/3 of its time; and (2) the optimum ±5°C, which includes the temperature a fish in the laboratory chooses to spend all of its time, if such temperatures are available (Magnuson et al., 1979; Crowder and Magnuson, 1983). Hokanson (personal communication) has compiled data suggesting that stream fishes spend ~90% of their time within the thermal range of optimum ±5°C. In some cases we estimated the thermal habitat for the entire year, in others only from June 5 to September 2 to be able to use the fishery yield prediction models of Christie and Regier (1988). In some cases, the thermal niche was measured in terms of depth days or the number of meters of the water column within the thermal niche summed across dates and expressed in terms of m-days of suitable space in the water column. For southern Lake Michigan, the size of the thermal niche was estimated in terms of the volume days of suitable thermal habitat and expressed as hm 3.7 days. For walleye and yellow perch, we calculated the weekly ha of bottom area exposed to suitable temperatures based on the approach of Christie and Regier (1988).

The average and minimum weekly thermal niche for the summer period (weeks 23-35) was also calculated for some species by computing the weekly mean and minimum weekly depth-days or volume-days of suitable thermal habitat. This measure of habitat size provides insight to periods during which suitable thermal habitat may be most limited (Coutant, 1987a; Rudstam and Magnuson, 1985). In the central basin of Lake Erie, the mass of cold water is small and usually anoxic in summer, so we did not estimate any thermal habitat in Lake Erie for coldwater fishes.

Lake trout is an example of a coldwater fish with a lower optimum (10°C) (Stewart et al., 1983) than most coldwater fishes. It is a species which, at first thought, should be most negatively influenced by warming of its habitat. Coho salmon is perhaps a more representative example of a coldwater fish with its optimum of 15°C (Hewett and Johnson, 1987). Yellow perch provide our example of coolwater species with an optimum of 23°C, and largemouth bass serves as the warmwater fish with an optimum of 27.5°C (Hewett and Johnson, 1987). Other individual species were used as well: lake whitefish at 12°C, and walleye at 20°C (Christie and Regier, 1988).

The statistical model equations relating summer thermal habitat to fishery yields from Christie and Regier (1988) are:

Lake trout	$\log_{10} SY = 0.807 \log10 THV + 0.944,\ r^2 = 0.86;$
Lake whitefish	$\log_{10} SY = 0.376 \log10 THV + 3.606,\ r^2 = 0.69;$
Walleye	$\log_{10} SY = 0.872 \log10 THA + 0.389,\ r^2 = 0.72$

where SY is an estimate of sustained yield in kg/year, THV is summer thermal habitat (±2°C) volume in $hm^3 \cdot 10$ days from June 3 to September 2, and THA is summer thermal habitat (±2°C) bottom area in ha. 10 days from June 3 to September 2.

The summer period as defined by Christie and Regier (1988) omits habitat that might exist during the spring and fall. Sufficient data for spring and fall, necessary for their statistical analyses, were not available; hence, their possible contribution to production could not be assessed. During the fall and late spring, there are periods during which water temperatures in their study lakes were very likely within the optimal ranges for lake trout and lake whitefish. The thermal habitat for these species during these periods may have contributed to the sustained yields they used in their analyses.

Scenarios

Thermal structure throughout the year was obtained from McCormick (1988) for Lake Michigan and from Blumberg (1988) for Lake Erie. OSU-simulated thermal structure was not available for Lake Erie. The three simulations we used are BASE, the present conditions ($1xCO_2$); the OSU scenario from the OSU $2xCO_2$ general circulation model; the GISS scenario derived from the GISS $2xCO_2$ general circulation model; and the GFDL scenario derived from the GFDL $2xCO_2$ general circulation model. For Lake Erie we used the profiles generated from 1975 (a cool year) and 1970 (a warm year).

Results

A profile of the thermal habitat of lake trout throughout the year (Figure 2) clarifies the concept of thermal niche as the m-days of water within ±2°C (black) and ±5°C (grey) of optimum. In these terms the thermal habitat under base climate is 1,500 m-days for ±2°C and 5,322 m-days for ±5°C. Typically the temperature in winter and spring mixis are too cold to be in the optimum range for lake trout as are the deeper waters in summer. In summer the near-surface waters are too warm. During fall mixis, much of the upper 100 m is within their thermal niche.

Under all three climate warming scenarios (OSU, GFDL, and GISS) the quantity of suitable annual thermal habitat increases greatly for lake trout (Figure 2). The increases are most pronounced for the GFDL and GISS scenarios where much of the upper 100 m of the water column reaches temperatures above 5°C in winter. Temperatures between 5 and 15°C persist long into the fall after climate warming. The size of

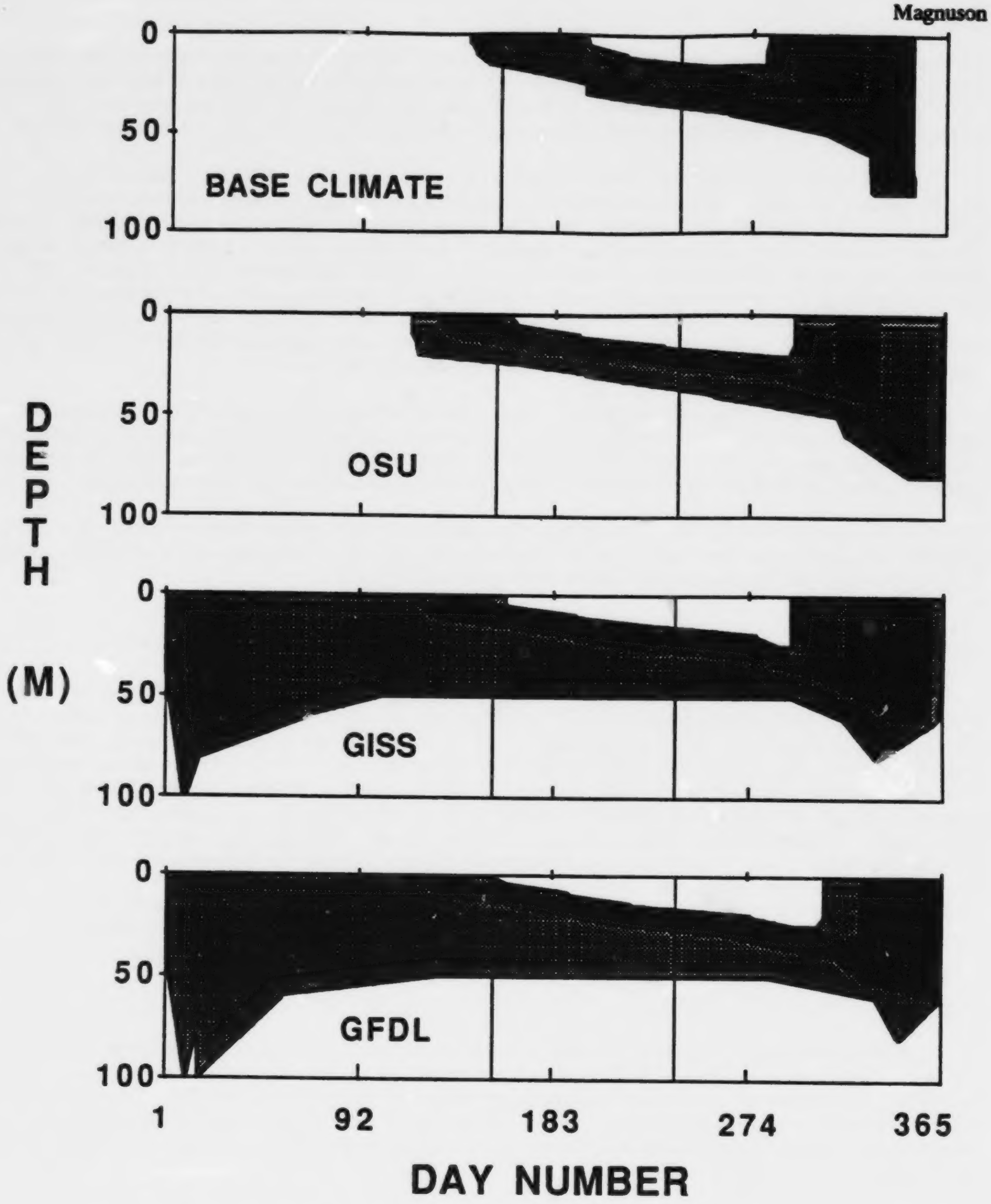

Figure 2. Extent of favorable thermal habitat before and after climate warming for a coldwater fish (lake trout) in southern Lake Michigan based on 10°C±2°C (black) and 10°C±5°C (grey). Vertical lines span the period (June 5 to September 2) used in the fishery yield prediction models of Christie and Regier (1988). Thermal scenarios for Lake Michigan provided by McCormick (1988).

unsuitably warm waters near the surface in summer increases with the climate warming scenarios. With the GISS and GFDL scenarios, about 40% of the upper 100 m are within the thermal niche of lake trout compared with 22% with the OSU scenario and only 14% under the base climate. Climate warming scenarios would increase the amount of favorable thermal habitat over the year even for a coldwater fish like the lake trout.

The size of the annual thermal habitat (Figure 3) not only increases for lake trout (factor of 1.5 to 2.7 x BASE) in Lake Michigan's southern basin under climate warming, but also for other coldwater fishes like coho salmon (1.3 to 1.5 x BASE), and coolwater fishes like yellow perch and walleye (1.4 to 3.1 x BASE). A small amount of thermal habitat also is opened up or expanded for warmwater fishes. In Lake Erie, climate warming increases the thermal habitat both for coolwater (1.2 to 2.3 x BASE) and warmwater (2.3 or greater x BASE) species (Figure 3). These results are robust and not sensitive to the particular method of estimation. The change in the amount of thermal habitat for lake trout measured in various ways (Figure 4) indicates that when the whole year is included, thermal habitat increases with climate warming regardless of whether ±2°C or ±5°C is used as a criterion or whether depth days or volume days are used as a metric.

The average summer habitat within the thermal niche of various fishes in Lake Michigan remains about the same or decreases (Figure 5). For the lake trout (coldwater) and the yellow perch (coolwater), the summer niche space decreases slightly as delineated by ±2°C of optimum and the vertical lines in Figure 2. Decreases (Figure 5) with climate warming for walleye (coolwater) and lake whitefish (coldwater) were large.

Sustained yields for Lake Michigan estimated by the Christie and Regier (1988) model are predicted to decrease only slightly for lake trout (2 to 6%) and lake whitefish (7 to 10%) with GISS and GFDL climate scenarios but increase significantly for walleye (+29 to +33%) (Figure 6).

HEADWATER STREAMS

Waters of many headwater streams in the Great Lakes Basin, especially those which originate in sandy and gravelly moraines with copious groundwater flows and springs, are inhabited by brook trout, a coldwater fish. Groundwater temperatures track local average air temperatures and are expected to increase with climate warming (Meisner et al., 1988). The major impact of climate change on the streams may be on the temperature of the groundwater entering the streams and on the rate of warming of water flowing downstream from headwater springs. A calibrated hydrometeorological model of stream temperature (Delay and Seaders, 1966) can be used to estimate summer stream temperatures before and after applying global climate change scenarios. Longitudinal distributions of brook trout in nondegraded streams normally are bounded between 19 and 24°C (Hawkes, 1975; Ricker, 1934), but Barton et al. (1985) and Bowlby and Roff (1986) found that water temperatures near 22°C marked the downstream habitat boundaries for Ontario. Combining the stream temperature estimates along the course of a stream with the thermal requirements of brook trout can provide estimates of changes in the size of thermal habitat for coldwater fishes in headwater streams.

Methods

A hydrometeorological model of summer thermal regime was implemented for the brook trout zone of two first-order headwater streams: the Rouge and Humber Rivers, which ultimately empty into Lake Ontario east of Toronto (Figure 1). The streams originate as groundwater discharge at about 10°C. The Humber River is about 5 km long with a mean discharge of .06 m^3/s and has forested banks, while the Rouge River is about 3 km long, has a mean discharge of .075 m^3/s, and flows through open fields. The streams are similar to other brook trout streams of southern Ontario.

The basic model (Delay and Seaders, 1966) simulates the change in temperature of the water as it flow downstream:

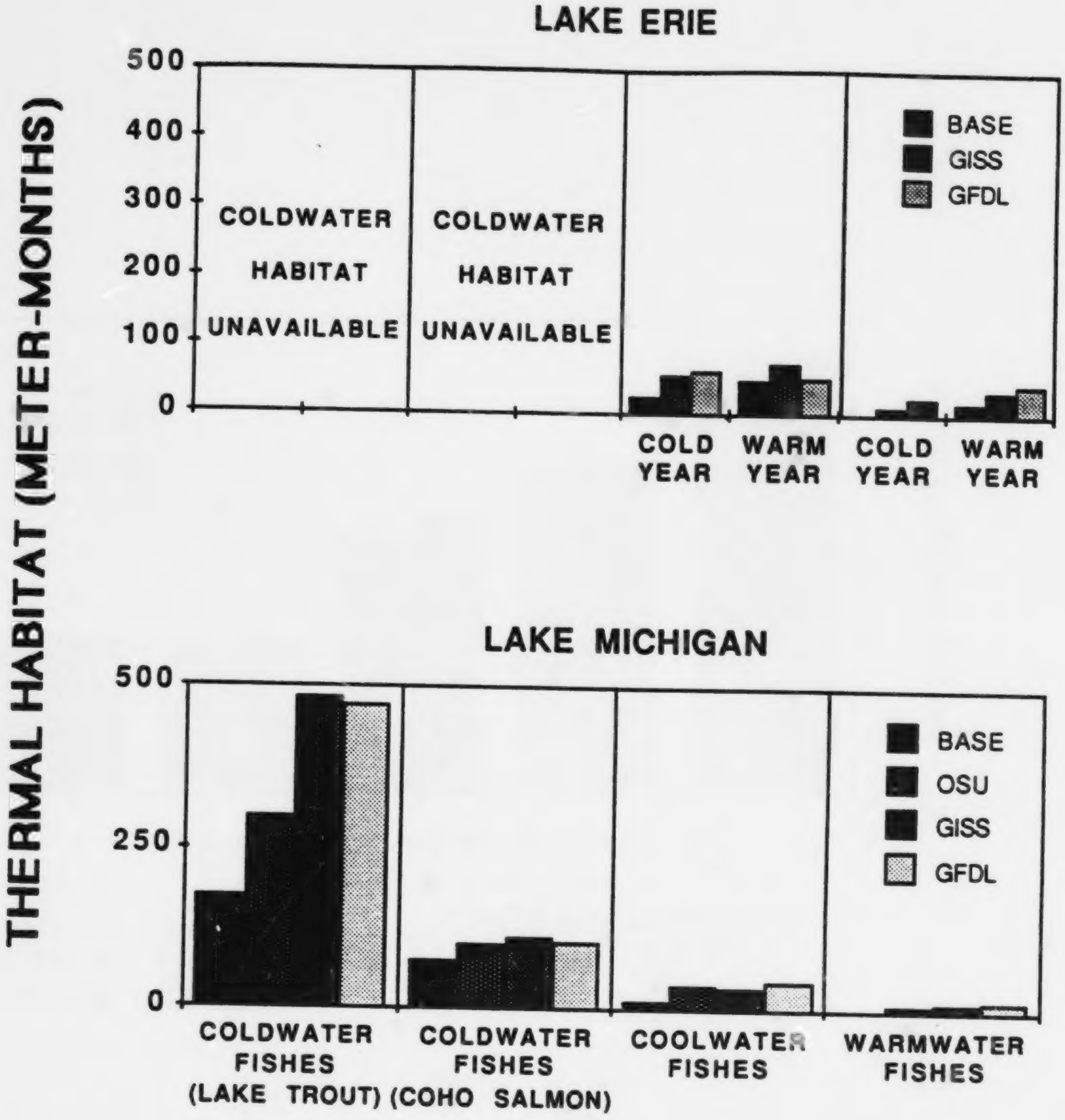

Figure 3. Amount of thermal habitat (optimum ±5°C) in the water columns of the central basin of Lake Erie and the southern basin of Lake Michigan for coldwater fishes (lake trout, coho salmon), coolwater fishes (yellow perch) and warmwater fishes (largemouth bass) for the BASE climate and for the OSU, GISS, and GFDL climate change scenarios using the thermal structure simulated by Blumberg (1988) for Lake Erie and by McCormick (1988) for Lake Michigan. For Lake Erie, the cold year was 1975 and the warm year was 1970.

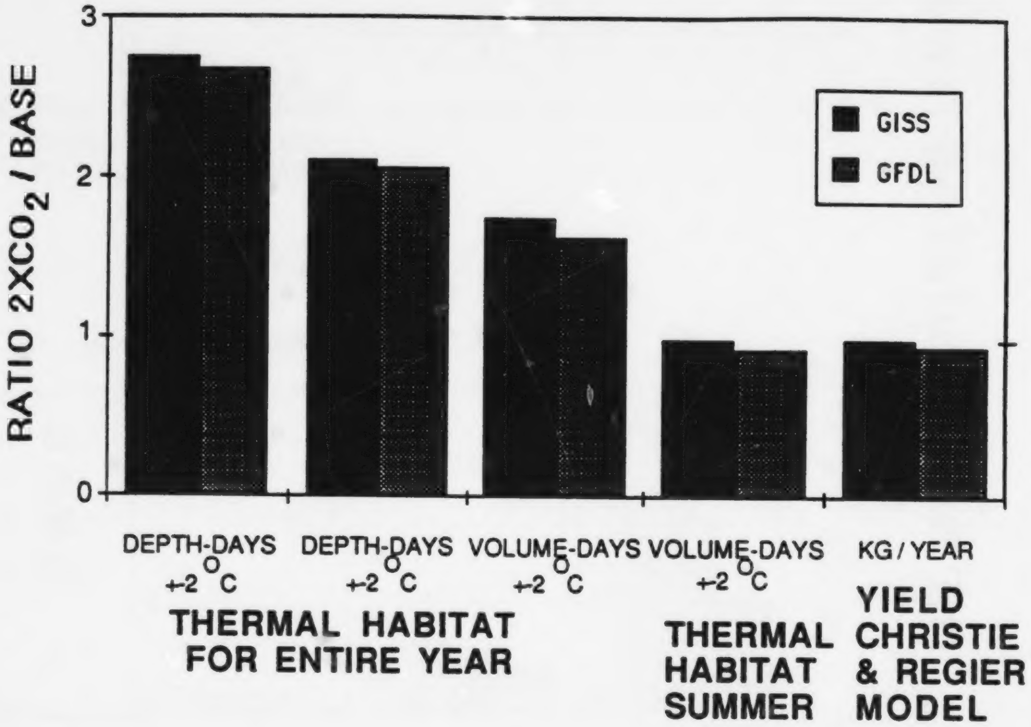

Figure 4. Change in the size of southern Lake Michigan thermal habitat expressed as a ratio of the GISS or GFDL estimates to the estimates from the BASE climate for a coldwater fish (lake trout) measured by various criteria; optimum ±2°C for the entire year or ±2°C for the summer months without and with the correction for lake hypsometry, and the estimated change in maximum sustained yields using the summer months and ±2°C criteria with the Christie and Regier (1988) model. The temperature structure of Lake Michigan is from McCormick (1988).

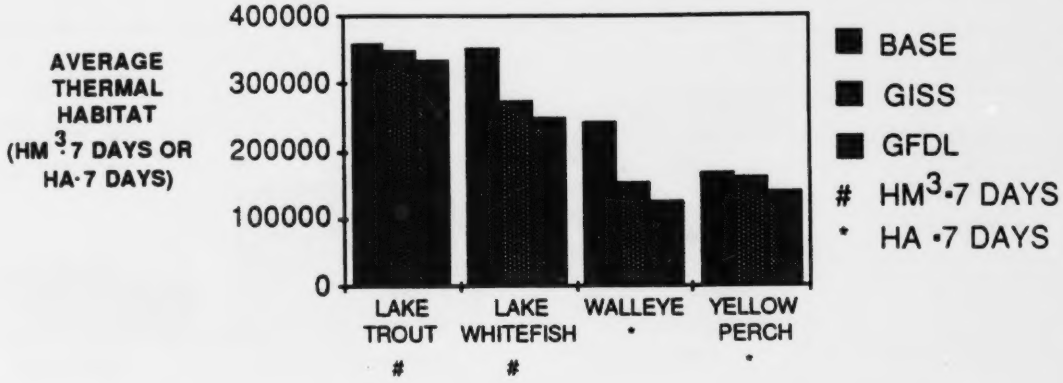

Figure 5. Average size of thermal habitats averaged over the summer months (weeks 23-35) for four Lake Michigan fishes based on a thermal niche of optimum ±2°C for the BASE climate and GISS and GFDL climate warming scenarios. Temperature structure of southern Lake Michigan is from McCormick (1988), hypsometry is for all of Lake Michigan.

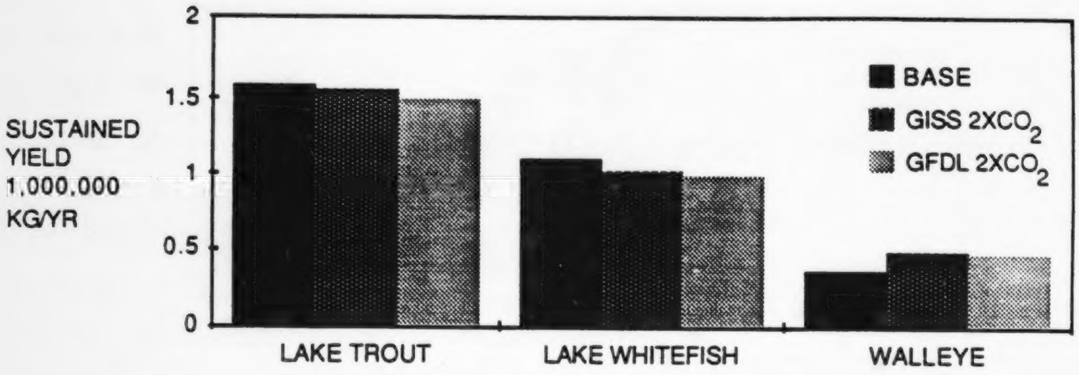

Figure 6. Maximum sustained yields of lake trout, lake whitefish, and walleye in southern Lake Michigan based on summer optimum ±2°C thermal habitat volumes for the BASE climate and the GISS and GFDL climate change scenarios. Yields estimated from the Regier and Christie (1988) model, thermal structure for southern Lake Michigan from McCormick (1988), hypsometry for the whole lake.

$$\Delta T_w = \frac{(SA)+v(t_v - T_w)}{V} \qquad \text{where,}$$

ΔT_w = change in temperature of parcel of water;
T_w = initial temperature of parcel of water;
A = surface area of parcel of water;
V = volume of parcel of water;
v = volume of inflow (e.g. groundwater, tributary);
t_v = temperature of inflow;
and S = change in energy stored in parcel of water.

S quantifies the energy flux across the air/water interface in a series of equations not shown here. This term treats the effects of air temperature, solar and longwave radiation, evaporation, conduction and wind on stream temperature explicitly. The model adapts to varying hydrological conditions along the stream and accounts for the thermal effects of shade.

The model was used to estimate the farthest downstream extent of temperatures suitable for brook trout in the warmest part of the diel cycle for a date in late July before and after applying the $2xCO_2$ climate warming scenarios. Observations indicate that the thermal barrier defining the downstream limit of the brook trout zone is 23°C in the Humber River and 24°C in the Rouge River.

Scenarios

Simulations of maximum July water temperatures were made using July 24, 1987, water temperatures from the sites and 1987 meterological data from the sites for BASE conditions and applying the $2xCO_2/1xCO_2$ adjustment statistics for July and August for the OSU, GISS, and GFDL models. Groundwater temperatures in the $2xCO_2$ simulations were derived using the mean annual air temperature for each scenario.

Results

The length of stream suitable for brook trout declines markedly in the $2xCO_2$ simulations of the Rouge and Humber Rivers (Figure 7). The loss of thermal habitat is least pronounced in the OSU scenario and most significant in the GFDL scenario, where suitable length in both streams is about half that found in BASE conditions. Although changes in groundwater discharge were not included in these simulations, any drying of the climate would likely exacerbate the habitat losses displayed here.

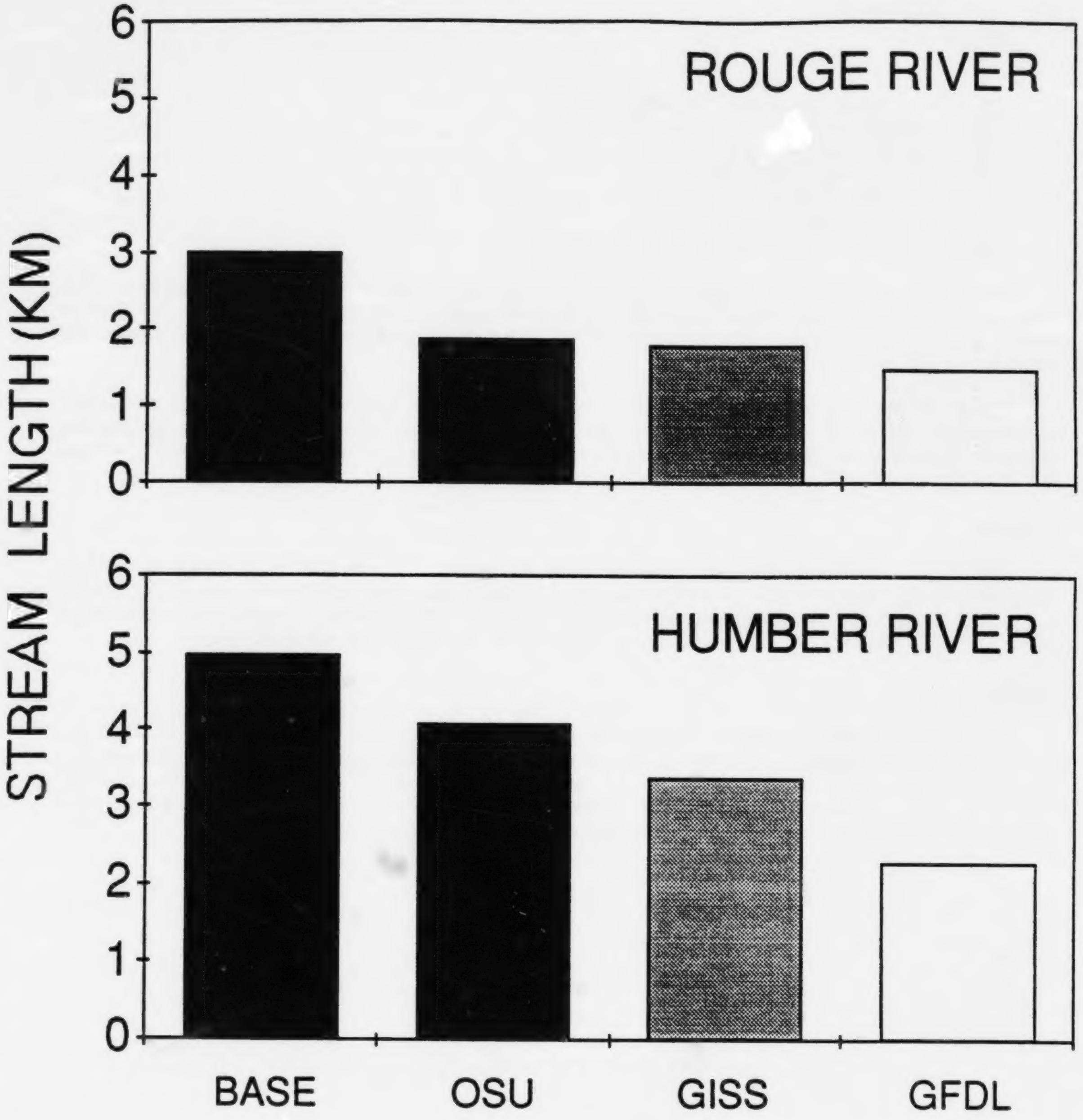

Figure 7. Length of the brook trout zone in two river headwaters in southern Ontario during July for the BASE climate and the OSU, GISS, and GFDL climate warming scenarios.

CHAPTER 3

GROWTH AND FOOD CONSUMPTION

The physiological processes of fishes depend strongly on water temperature (Fry, 1971), and coupled with the rate of food consumption temperature determines the rate at which fish grow. The fish bioenergetics model of Kitchell et al. (1977) as given in Hewett and Johnson (1987) allows the computation of growth given the food consumption and the water temperature environment of the fish. Alternatively prey consumption can be estimated from knowledge of water temperature and growth using the bioenergetic model. This methodology has been applied to a variety of individual fish species (Kitchell et al., 1977; Rice et al., 1983; Stewart et al., 1983; Rice and Cochran, 1984; Stewart and Binkowski, 1986) as well as of interspecific interactions (Kitchell and Breck, 1980; Stewart et al., 1981, Lyons and Magnuson, 1987).

The impact of increased water temperatures on fish growth and food consumption in the Great Lakes will be assessed using the bioenergetic models. The potential effects of a temperature change on growth and food consumption are estimated by applying the model to temperature scenarios representing conditions before and after climate warming. Simulations using different proportions of the maximum consumption rate were used in an attempt to bracket a range of potential energetics responses to climate warming.

METHODS

The bioenergetics model computes energy budget statistics for individual fish using the equation $G=C (R+F+U)$, where G is the growth rate, C the rate of food consumption, R the sum of metabolic costs, and F and U the waste products due to egestion and excretion (Kitchell et al. 1977). All of the terms in this equation are strongly dependent on temperature. The physiological processes (R, F, and U) are known functions of temperature and feeding rate. Therefore, fish growth may be computed if the temperature and feeding rate are known.

To evaluate the range of potential energetics responses of fish to climate warming, two sets of energetics simulations were run using different assumptions about prey consumption rates. The first set of simulations were run under the assumption that as the climate changes, the fish will continue to feed at the same fraction of their maximum consumption rate as in the base period. The assumption that the proportion of maximum consumption (called the "P-value") is constant implies that prey availability will increase with climate warming at a rate such that actual consumption remains the same fraction of maximum potential consumption. P-values were chosen to produce growth in the range observed in the Great Lakes for baseline climatic conditions. Baseline P-values were 0.7 for largemouth bass, 0.45 for yellow perch, and 0.75 for lake trout. A P-value of 0.7 means, for example, that a fish consumes prey at a rate of 0.7 times the maximum consumption rate, which is a function of temperature and body weight. In the second set of simulations, annual prey consumption was assumed to remain constant at baseline levels in the $2xCO_2$ scenarios. This assumption means that the P-value in the $2xCO_2$ simulations is lower than in the base scenarios.

In both sets of simulations, growth was computed for representative "cold," "cool," and "warm" water fishes: lake trout, yellow perch, and largemouth bass. Each individual simulation consisted of a yearling fish exposed to one year of the water temperature scenarios, beginning in February, the coldest month in the annual cycle. Starting weights were 8 g for yellow perch and 20 g for both lake trout and largemouth bass. For all species, a generic prey was used in the models with an energy density corresponding to a mixed diet of fish and invertebrates. The lake trout prey was given a higher energy density to reflect the importance of the energy-rich alewife in their diet (Stewart et al., 1983).

An important assumption used in our bioenergetic simulations is that fish behaviorally thermoregulate; they seek out a preferred temperature which is close to their optimal temperature for growth (Neill and Magnuson, 1974; Magnuson et al., 1979). Thus, the maximum water temperature in our scenarios was limited to the preferred temperature of the species being modeled. The assumption of behavioral thermoregulation includes the implicit assumptions that the preferred temperature is available to the fish, that oxygen concentration is adequate at that temperature, and that sufficient prey are available to support the assumed feeding rates. The following preferred temperature values were used: 10°C for lake trout, 23°C for yellow perch, and 27.5°C for largemouth bass (Hewett and Johnson, 1987).

SCENARIOS

The energetics simulations with prey consumption held constant at baseline levels in $2xCO_2$ climate conditions were run using McCormick's (1988) water temperature scenarios for the southern basin of Lake Michigan (see Figure 1). The simulations with prey consumption increasing with climate warming were applied to three Great Lakes near-shore sites representative, respectively, of "cold," "cool," and "warm" thermal environments in baseline climate conditions: Thunder Bay, Ontario (Lake Superior), Evanston, Illinois (Lake Michigan), and Put-In Bay (Lake Erie) (see Figure 1). EPA's standard procedure was followed to create air temperature scenarios for the three sites. Daily average air temperature observations from Cleveland, Ohio (Put-In Bay site), Chicago, Illinois (Evanston site), and Thunder Bay, Ontario (Thunder Bay site) were used to compute monthly average air temperatures. These BASE scenarios covered the period 1951-1980 for Chicago and Cleveland, and 1953-1980 for Thunder Bay. The OSU, GISS, and GFDL air temperature scenarios were created by multiplying temperatures in the base period by the appropriate $2xCO_2/1xCO_2$ ratio from the climate model output, using the closest model gridpoint to each site of interest. The air temperature scenarios were then used to create water temperature scenarios representing inshore epilimnetic conditions at each site. Multiple regression models were developed relating the observed monthly averaged water temperature to the current month's air temperature and the previous month's air and water temperatures (see Table 1 for equations). Three different seasonal models were developed for each site: a winter model and models for the periods of rising and falling water temperatures. Water temperature data for Put-In Bay were obtained from the Put-In Bay fish hatchery. The years 1954-1973 and 1976 were used to calibrate the model. The city of Evanston water supply intake was the source of Lake Michigan temperature data, spanning the period 1974-1987. Lake Superior data between 1958 and 1977 were obtained from the city of Thunder Bay water intake. For each site, daily water temperatures were averaged to monthly values.

The base period water temperature scenarios (BASE) were derived by applying the regression models to the monthly average observed air temperatures for 1951-1980. The model was run in an iterative fashion, using the predicted water temperature for the current month in the computation of the next month's water temperature. This approach requires in addition to the air temperature data only an initial guess of the water temperature for January 1951 to generate the entire 30-year scenario. Similarly, the OSU, GISS, and GFDL water temperature scenarios were derived by applying the regression equations to the respective air temperature scenarios. Water temperature scenarios were then modified for thermoregulation by cutting off the temperature at the fishes' preferred value.

The water temperature scenarios are summarized in Figure 8. The left column displays the average and extreme values for the water temperature scenarios over the entire 30-year period. For every site, the GFDL climate change scenario is the warmest, the GISS scenario intermediate, and the OSU scenario the coolest. In each case, Lake Erie experiences the greatest increase in annual mean temperature, while Lake Superior changes the least. The GFDL mean annual water temperature is 3.3°C warmer than the BASE scenario for the Lake Superior site, 3.4°C warmer for Lake Michigan, and 5.8°C warmer for Lake Erie. The seasonal plot of average monthly temperatures in the right column of Figure 8 shows that the GFDL scenario is uniformly warmer than the OSU and GISS scenarios, with the largest difference occurring in midsummer. The horizontal lines bisecting the annual temperature cycle are the optimal temperatures for growth used in the bioenergetics simulations. Under the behavioral thermoregulation assumption, the fish follow these lines as soon as the near surface waters warm above their preferred temperature. The increased width of the thermoregulation lines for the OSU, GISS, and GFDL scenarios over those for the BASE scenario means that the fish have a longer growing season wherein they may, with behavioral thermoregulation, operate in optimal temperature conditions for growth.

Table 1. Regression Equations Relating Water Temperatures to Air Temperatures for Warm, Cool, and Cold Great Lakes Sites[a]

Thunder Bay (Lake Superior)

Winter: $WT=0.418+0.312WTL+0.036AT-0.023ATL$ $R2=0.38$

Warming: $WT=0.440+0.157WTL+0.357AT+0.239ATL$ $R2=0.90$

Cooling: $WT=2.534+0.324WTL+0.164AT+0.174ATL$ $R2=0.91$

Evanston (Lake Michigan)

Winter: $WT=1.475+0.363WTL+0.106AT-0.027ATL$ $R2=0.57$

Warming: $WT=3.347+0.304WTL+0.217AT+0.208ATL$ $R2=0.90$

Cooling: $WT=1.928+0.377WTL+0.382AT+0.067ATL$ $R2=0.96$

Put-In Bay (Lake Erie)

Winter: $WT=0.959+0.154WTL+0.140AT-0.053ATL$ $R2=0.49$

Warming: $WT=2.896+0.451WTL+0.524AT-0.009ATL$ $R2=0.96$

Cooling: $WT=-7.642+1.022WTL+0.386AT-0.173ATL$ $R2=0.94$

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[a] Symbols and units: WT = current month's mean water temperature; WTL = last month's mean water temperature; AT = current month's mean air temperature; ATL = last month's mean air temperature (all temperatures °C); R2 = coefficient of determination.

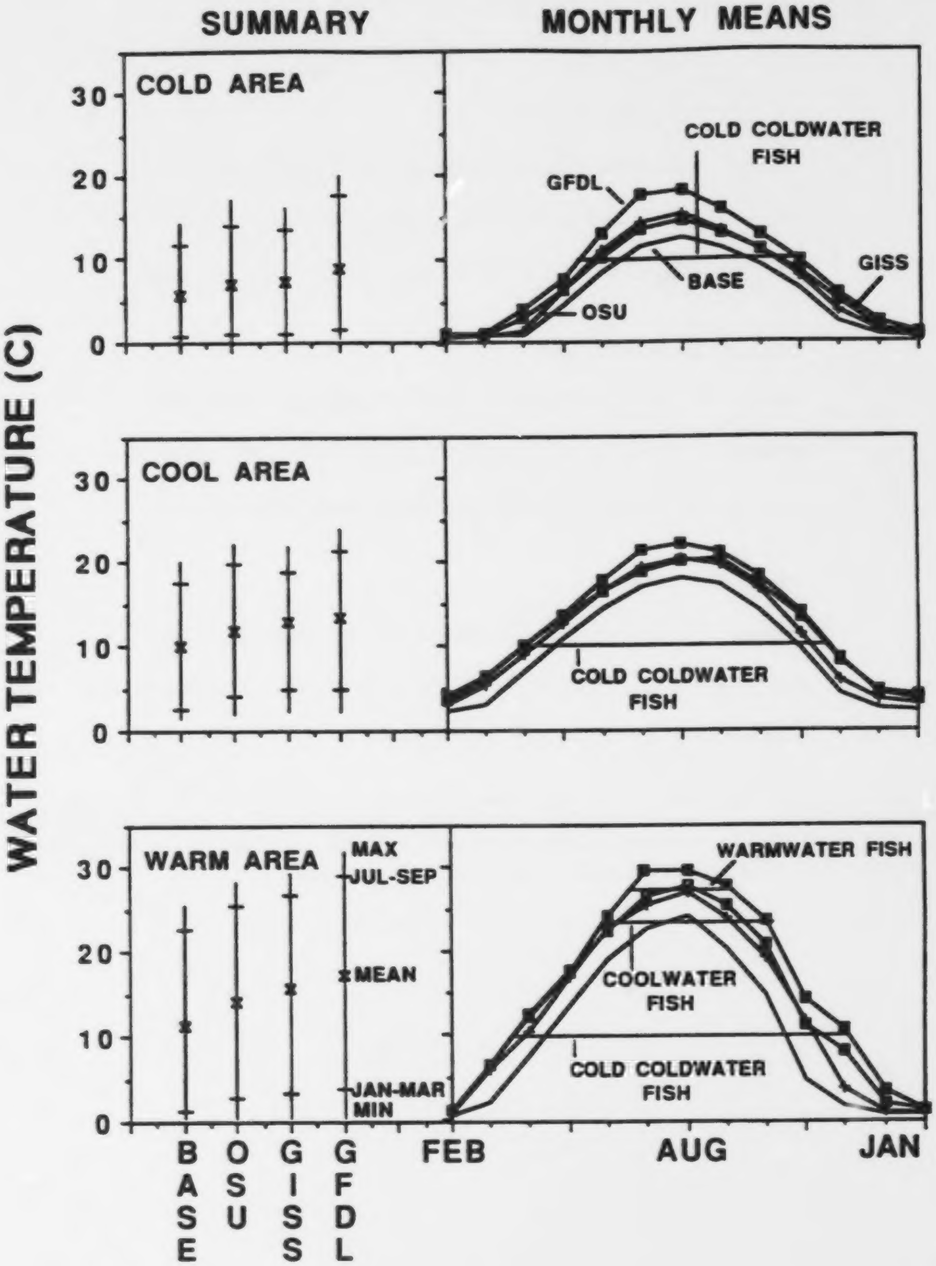


Figure 8. Water temperatures in cold, cool, and warm neritic areas of the Great Lakes (Thunder Bay, Lake Superior; southern Lake Michigan; Put-In Bay, Lake Erie) for the BASE climate and the OSU, GISS, and GFDL scenarios based on multiple regression relations between water intake and air temperatures. Optimum temperatures for three thermal guilds of fishes are shown.

Water temperatures in the base scenario for Evanston, Lake Michigan, are several degrees cooler in midsummer than those found for the central Lake Michigan site by McCormick (1988). This difference is likely owing to the fact that the water intake site at Evanston is in a nearshore site subject to rapid temperature drops caused by upwelling events, internal waves, etc. These factors may also cause our scenarios at Thunder Bay and Put-In Bay to underestimate actual epilimnetic temperatures.

RESULTS

The consequences of longer growing seasons under the $2xCO_2$ climate scenarios when prey consumption is allowed to increase with climate warming are readily apparent in Figure 9. Coldwater and coolwater fishes (lake trout and yellow perch) exhibit clear increases in annual growth over that in the base period. The ratio of growth of cold coldwater and coolwater fishes in $2xCO_2$ conditions to that in the base scenario ranges from 1.2 to 1.7 (mean 1.3). Warmwater fishes (largemouth bass) cannot grow in any of the cold area temperature scenarios or in base conditions in the coolwater area. They experience minor growth in the $2xCO_2$ scenarios in the cool area, and large increases in growth in the warm area ($2xCO_2/1xCO_2$ ratios of 3.2, 4.3, and 5.7). Although in all cases the fishes under the GFDL scenarios grow faster than those under the GISS scenarios, differences are minor compared with differences between the $2xCO_2$ and baseline scenarios.

For all three thermal guilds of fishes, the site with the largest simulated growth is the warm area. This may seem inconsistent with the intuitive notion that a coolwater fish should do best in a coolwater environment. However, Figure 8 shows that the water temperature scenarios for the warm area possess longer periods when preferred temperatures may be available for all three fish species. In the case of Lake Erie lake trout, suitable temperature and oxygen conditions are not found in the vicinity of the Put-in Bay site in summer. Figure 8 shows that Put-In Bay is potentially habitable for lake trout for 3-4 months during winter after climate warming. Historically, lake trout in Lake Erie have used the eastern basin as a summer refuge and migrate into the central basin during the fall. Also, other coldwater fishes (cisco and lake whitefish) use areas near Put-In Bay in fall for spawning, but are not present during summer.

Results for prey consumption (Figure 10) closely parallel those for growth, because growth is an integrated form of consumption. Changes in consumption between the BASE and $2xCO_2$ scenarios ranged from factors of 1.2 to 1.7 (mean 1.3) for cold coldwater and coolwater fish. Warmwater fish consumption rates ranged from 1.6 to 2.8 (mean 2.1) times higher than the base period. Similar to the result seen for growth, Lake Erie fishes consumed more than their counterparts in the other lakes.

The effect of the two assumptions about prey availability on fish growth following climate warming is readily apparent in Figure 11. When prey consumption is allowed to increase with climate warming, growth increases. The $2xCO_2/1xCO_2$ growth ratios are about 1.6 for coldwater fish and 1.4 for coolwater fish. Warmwater fish, which lost weight in the baseline scenario, are able to grow in the $2xCO_2$ scenarios. When consumption is held constant in the climate warming scenarios, growth for all three fish guilds declines. Warmwater fish lose even more weight than in the base scenario. Coolwater and coldwater fishes also suffer in the equal consumption simulations ($2xCO_2/1xCO_2$ growth ratios of 0.7, 0.9). Differences in growth caused by the two assumptions concerning prey consumption are much larger than the differences in growth between climate model scenarios. This implies that fish growth is more sensitive to changes in food web structure (Figure 12) than to differences in the temperature scenarios.

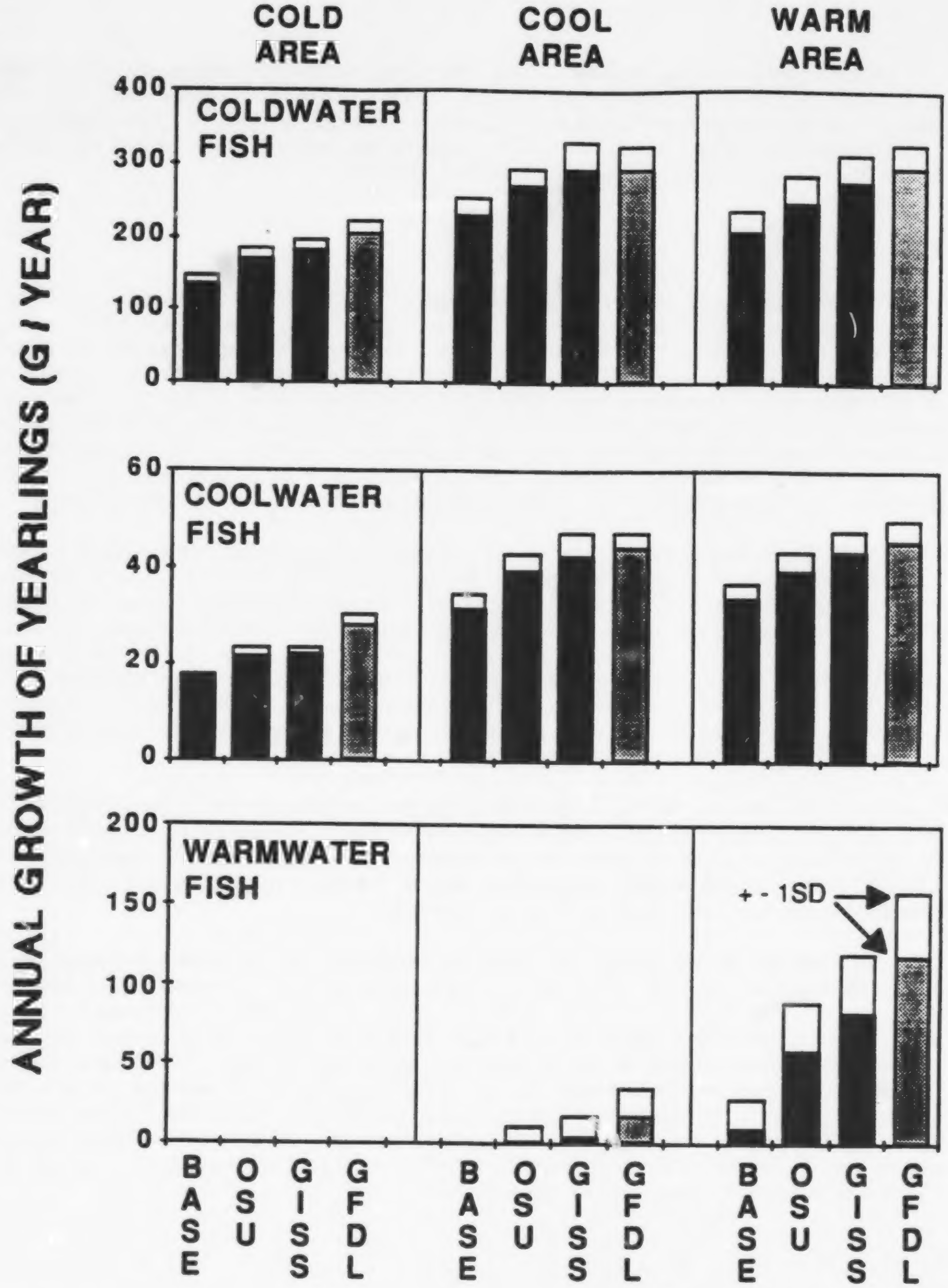


Figure 9. Annual growth (±1SD) of cold, cool, and warmwater fishes (lake trout, yellow perch, and largemouth bass) in cold, cool, and warm neritic areas of the Great Lakes (Thunder Bay, Lake Superior; southern Lake Michigan; Put-In Bay, Lake Erie) for the BASE climate and the OSU, GISS, and GFDL climate change scenarios. These simulations assume prey consumption rises with climate warming.

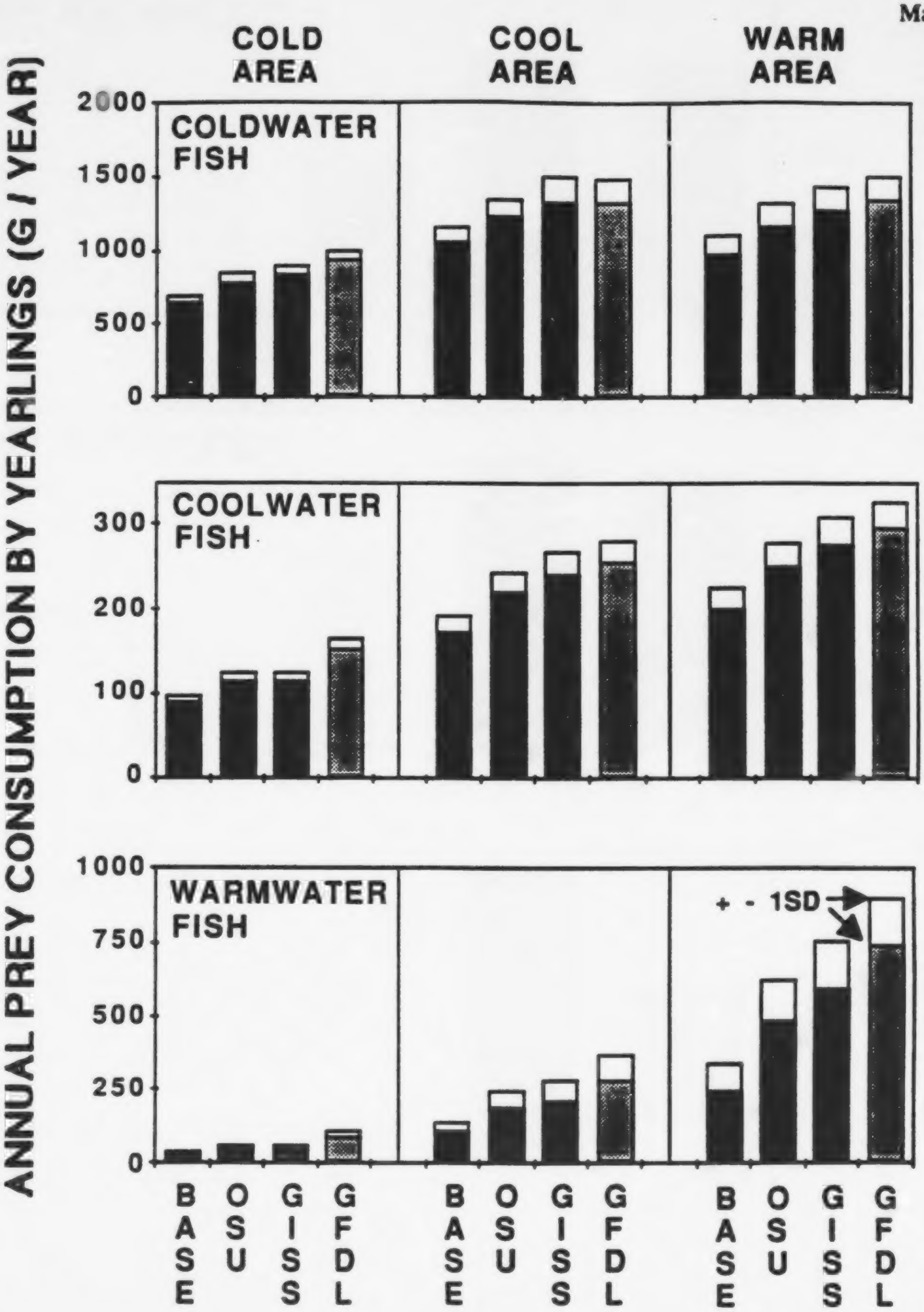


Figure 10. Annual prey consumption (±1SD) by cold, cool, and warmwater fishes (lake trout, yellow perch, and largemouth bass) in cold, cool, and warm neritic areas of the Great Lakes (Thunder Bay, Lake Superior; southern Lake Michigan; Put-In Bay, Lake Erie) for the BASE climate and the OSU, GISS, and GFDL climate warming scenarios.

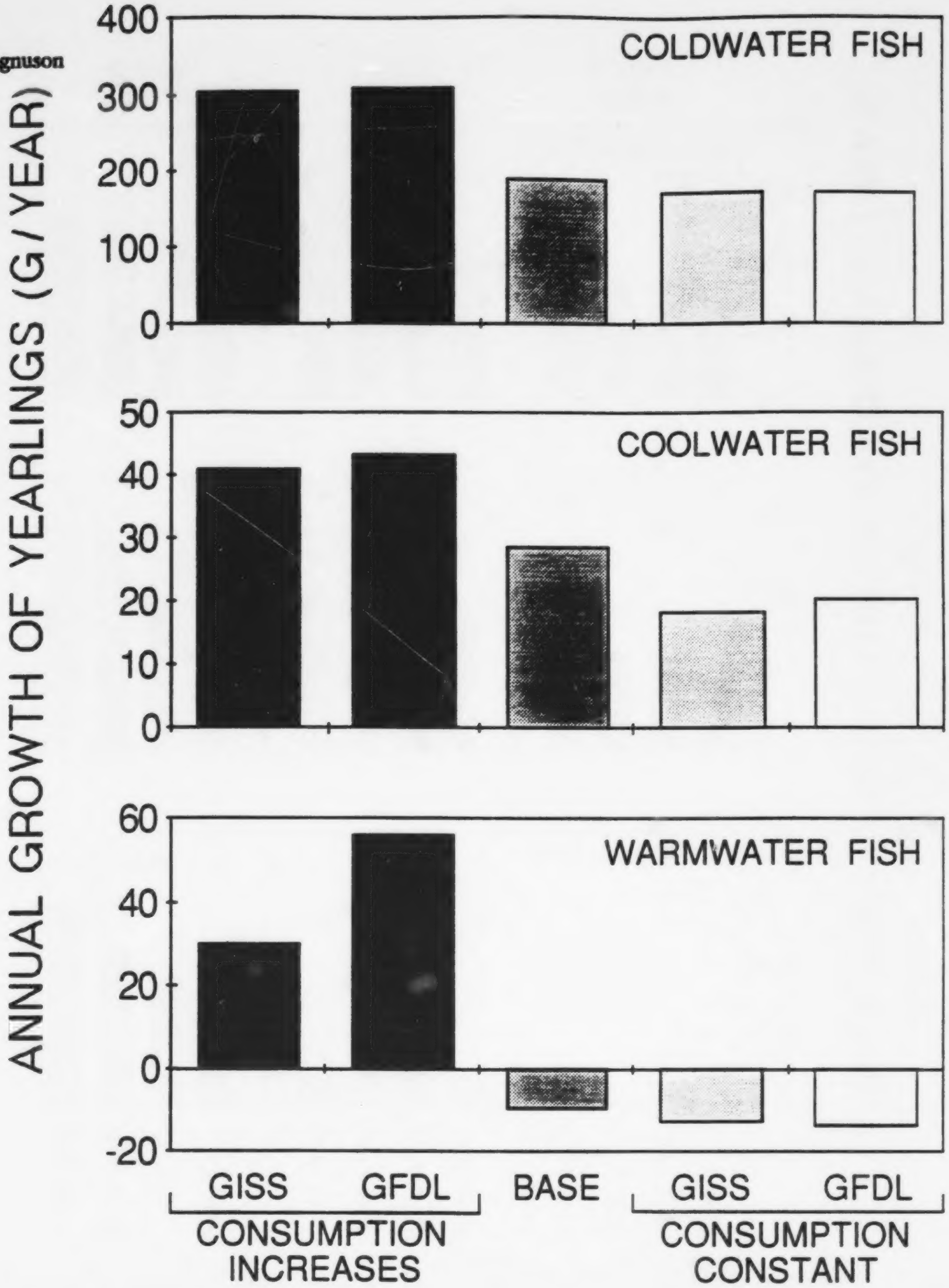
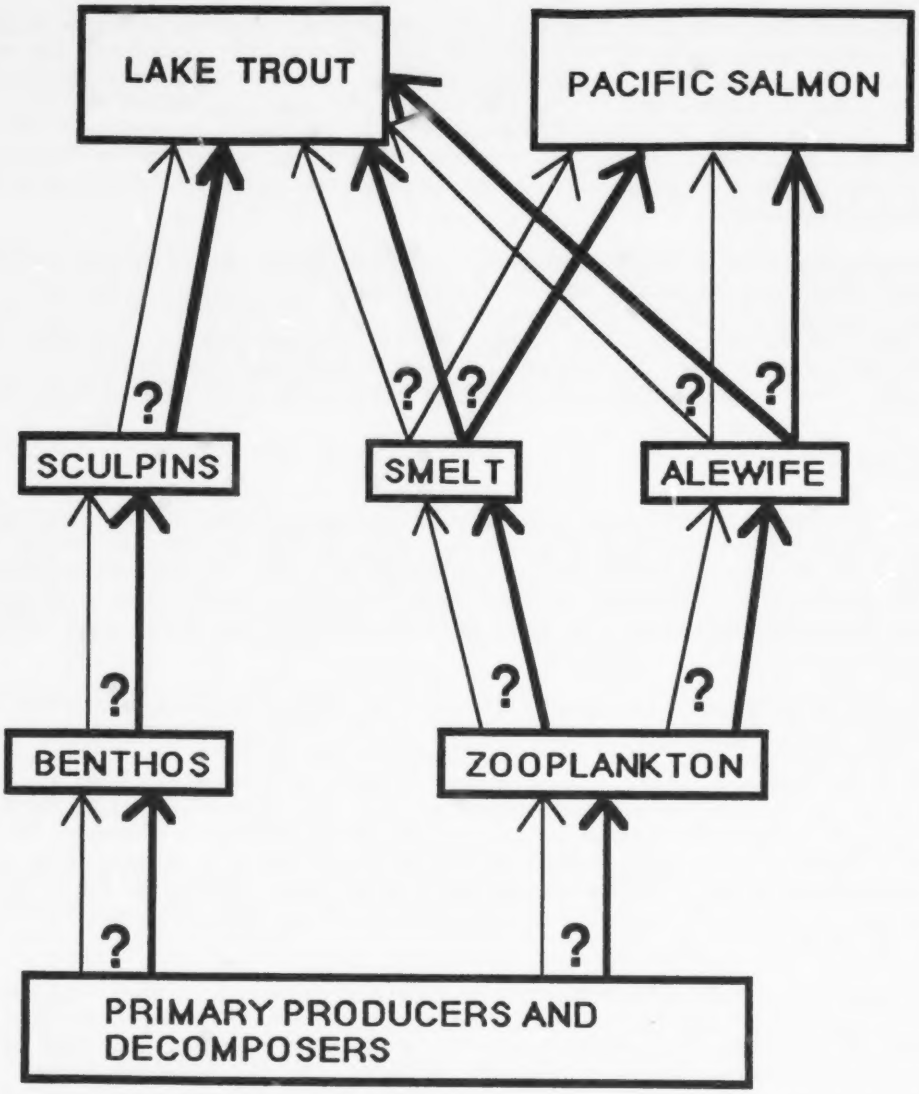


Figure 11. Annual growth of cold, cool, and warmwater fishes (lake trout, yellow perch, and largemouth bass) in southern Lake Michigan with prey consumption allowed to increase or stay constant as climate warms. Lake Michigan thermal scenario provided by McCormick.


LAKE TROUT
PACIFIC SALMON
SCULPINS
SMELT
ALEWIFE
BENTHOS
ZOOPLANKTON
PRIMARY PRODUCERS AND DECOMPOSERS
?
?
?
?
?
?
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?
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Figure 12. Idealized Great Lakes food web diagram. Question marks and varied widths of energy transfer arrows highlight the uncertainties about whether production at lower trophic levels will be sufficient to meet the predatory demand of higher levels.

CHAPTER 4

POPULATION RESPONSE

Smallmouth bass, a warmwater fish, was used to demonstrate population responses to global climate warming. These animals live in the shallowest, most protected areas of the Great Lakes and are, therefore, likely to show the effects of climate warming earlier than many other species. However, since the Great Lakes watershed is near the northern limit of the natural range of the species, these effects will almost certainly be positive. This must be kept in mind while reviewing our results. Our choice of smallmouth bass was dictated solely by the fact that reliable recruitment and growth models for this species were already in existence and could be used immediately to produce credible forecasts of the effects of climate change on year class strength and population abundance.

The reproductive rate of any fish population is strongly dependent on the average survival of the young and the age at which adults reach sexual maturity. For smallmouth bass, both survival and age of maturity are strongly dependent on temperature. Overwinter survival of young bass depends on their size at the end of their first summer, and this in turn depends on the length of the growing season (Shuter et al., 1980). Similarly, age of first maturity varies directly with growth rate and, therefore, with length of the growing season (e.g., Wrenn, 1980, 1984). Thus, population responses of these fishes should be affected by a change in climate.

METHODS

Three sites (Figure 1) were chosen for the estimated population response of smallmouth bass: warm -- a shallow bay near Long Point on Lake Erie; cool -- a shallow bay, Baie du Dore, Lake Huron; and cold -- a shallow bay near Thunder Bay on Lake Superior. Smallmouth bass populations presently exist at both the Lake Erie and Lake Huron sites. Data from these populations were used extensively in developing forecasts of the potential effects of climate change. The Lake Superior site is an imaginary bay physically similar to Baie du Dore.

The smallmouth bass recruitment model of Shuter et al. (1980) was used to estimate winter mortality and the length of the growing season for young-of-the-year fish. Growth for all age groups was estimated using the Kitchell bioenergetics model (Kitchell et al., 1977; Hewett and Johnson, 1987). Age of first maturity (Tm) was defined as the number of years of growth required to exceed a total weight of 260 g. This weight is representative of the size at first maturity among the males of both the Lake Erie and Lake Huron populations. It is also similar to the values reported for other smallmouth bass populations (Carlander, 1977). The food availability estimates used in the bioenergetics model were derived from the actual growth rates observed in the Lake Erie and Lake Huron populations in recent years. These estimates indicate that fish in these populations are currently feeding at levels well below satiation. We assumed that these food availability parameters were not affected by climate change.

For each site, the combined growth recruitment model was run using current water temperatures and using estimated water temperatures derived from GISS $2xCO_2$ air temperatures. The population effects produced by this change in climate were evaluated in terms of the following ratios: (average YCS GISS $2xCO_2$/ average YCS current) and (average adult abundance GISS $2xCO_2$/ average adult abundance current). These ratios were calculated using equations derived from the simple age-structured population model described below.

The model we use is constructed directly from the cohort and recruitment models described in Ricker (1975). Assume all population members aged 1 and over experience a constant instantaneous natural mortality rate equal to M and that all fish aged Tm and over experience a constant instantaneous fishing mortality rate equal to F. Assume a relation between YCS at age 1 (N_1) and the number of breeding adults (S) which has the following form:

$$N_1 = aS^{\beta} \qquad (1)$$

$$\text{where } S = \sum_{i=T_m}^{\infty} N_i \quad \text{and } \beta < 1.$$

Preliminary analyses of the Lake Opeongo smallmouth bass data base show that such an N1-S relation does hold for that population. At steady state, the following equations hold:

$$S = N_{T_m}\left(\frac{1}{1-e^{-Z}}\right) \qquad (2)$$

$$N_{T_m} = N_1 e^{-M\left(T_m - 1\right)} \qquad (3)$$

Now, by substituting equations 2 and 3 into 1, we obtain:

$$N_1 = a\left(\frac{N_1 e^{-M\left(T_m-1\right)}}{1-e^{-Z}}\right)^{\beta}$$

which simplifies to:

$$N_1 = a^{\frac{1}{1-\beta}}\left(\frac{e^{-M\left(T_m-1\right)}}{1e^{-Z}}\right)^{\frac{\beta}{1-\beta}} \qquad (4)$$

Now, assume that a in equation 4 is proportional to winter survival (w). Further, assume that Z, M and β are constant. Under these conditions, equation 4 requires that:

$$N_1 \sim \left(we^{-\beta M T_m}\right)^{\frac{1}{1-\beta}} \qquad (5)$$

Therefore, any change in first year winter survival (w) or age of first maturity (Tm) can be evaluated in terms of the resulting changes in both steady-state YCS (using equation 5) and S (by substituting 5 into 3 and then substituting the resultant equation for N_{T_m} into 2). Equation 5 shows that the only other parameters that affect the results are M and β. The level of fishing mortality is not important. All climate change scenarios that we explore here are based on values for M and β (0.3 and 0.65, respectively) derived from preliminary analyses of the long term behavior of the Lake Opeongo smallmouth bass population.

SCENARIOS

For both the Lake Erie (Ontario Ministry of Natural Resources, unpublished data) and Lake Huron (Shuter et al., 1985) sites, littoral temperature data were available from several locations within the overall habitat of resident populations of smallmouth bass. These data were used to construct annual temperature curves representative of the average temperatures experienced by members of these populations over the last 10-20 years. A current temperature curve for the Lake Superior site was derived using the Baie du Dore water temperature curve, Baie du Dore and Thunder Bay air temperatures, and the equations of Shuter et al. (1983) to translate observed air temperature differences into water temperature differences. The estimated GISS $2xCO_2$ effect on each water temperature annual cycle was determined by first determining the GISS-predicted change in annual average air temperature and then using that change with the Shuter et al. (1983) equations to estimate the resultant change in the annual water temperature cycle.

RESULTS

The smallmouth bass populations in shallow protected bay habitats experience an increase in reproductive success and fishable population size in cold, cool, and warm areas with a GISS $2xCO_2$ climate warming (Figure 13). As expected, the relative advantage is greater in the cold areas. The imaginary Lake Superior population probably is not viable prior to climate change but after climate change, it appears quite viable. The population in the warm area near Long Point, Lake Erie is already doing well and would neither gain nor lose from climate warming. The cool area near Baie du Dore, Lake Huron exhibits a positive response, with year class strength increasing by a factor of 9 and the fishable population by a factor of 23, both of which are intermediate between the response at the warm and cold sites.

The increases in reproductive success and population size occur in the model simulations because the length of the growing season increases, overwinter survival of young-of-the-year smallmouth bass increases, and the fish spawn at earlier ages (Figure 13). The growing season is simulated to increase with climate change by a factor of approximately 1.3, for both young-of-the-year and adults. Overwinter survival of young of the year fish increases from 10% to 88% in cold areas and nears 100% in cool and warm areas. Age of maturity decreases by 1 to 3 years, so generation time decreases.

Our forecasts for smallmouth bass were uniformly positive. The reader must keep in mind that similar detailed modeling for cool or coldwater species could well produce very negative forecasts. It was our intention that the smallmouth bass work fulfill two primary functions:

(1) provide a credible forecast, given our present state of knowledge, of the likely effects of climate change on a typical warmwater species currently resident in the Great Lakes drainage basin, and

(2) provide an example of the kinds of detailed models necessary to forecast the effects of climate change at the population level. Our smallmouth bass forecasts are in no way representative of the response expected from a "typical" species currently resident in the Great Lakes area.

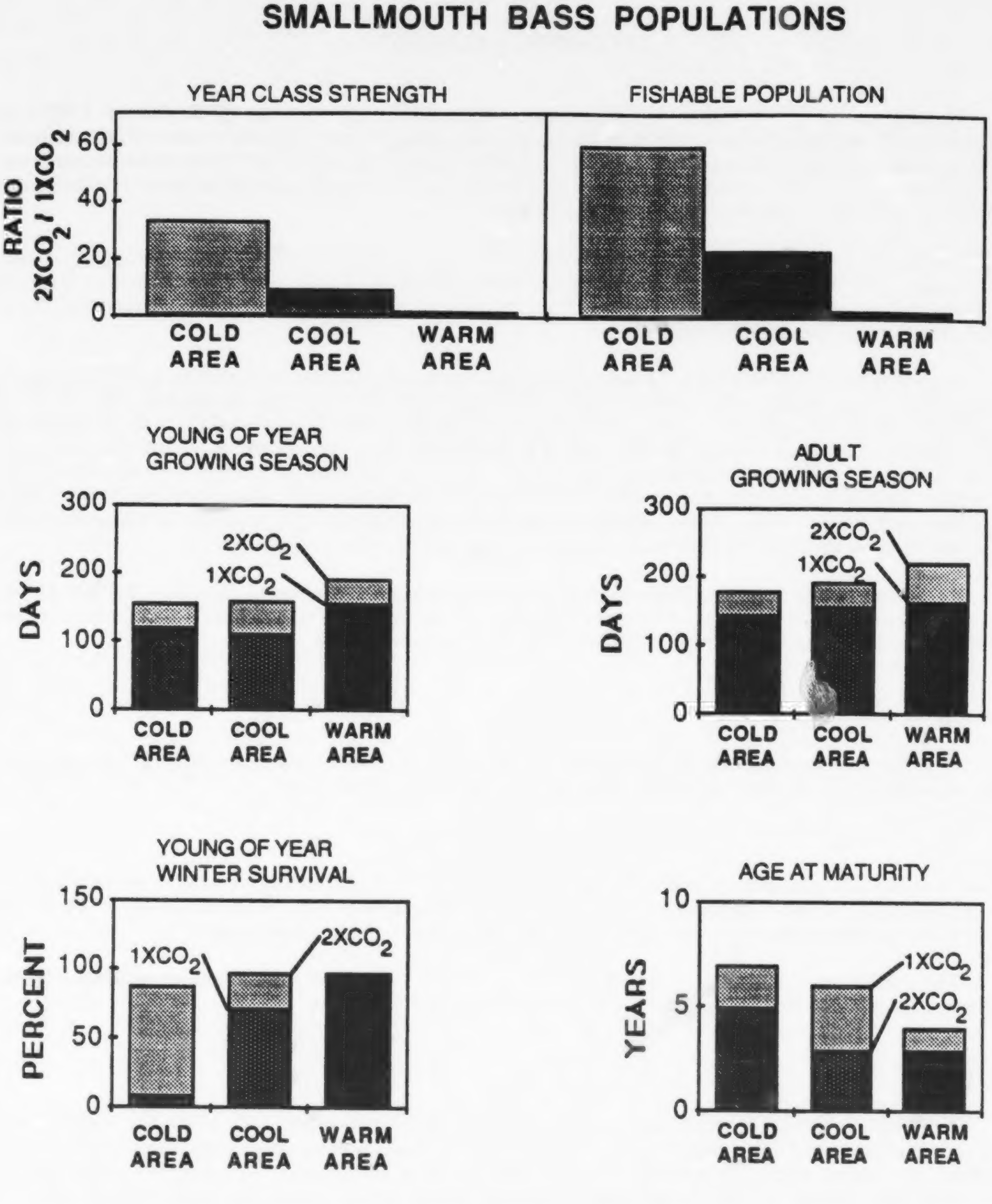

Figure 13. Changes in smallmouth bass populations with GISS climate warming in shallow protected littoral areas of Lake Superior (cold), Lake Huron (cool), and Lake Erie (warm).

CHAPTER 5

ECOSYSTEM RESPONSE

Richard A. Vollenweider (personal communication) reminded us of works by F. Ruttner (1931), A. Thienemann (1930), and other early German limnologists comparing the tropical inland waters of Java, Sumatra, and Bali and the temperate lake systems of Northern Europe. Lakes which appeared similar in all aspects except temperature differed with respect to relative trophic status, with the tropical lakes being more productive and appearing to be more eutrophic than the temperate lakes.

The rate of photosynthetic production in well-lit surface waters is dependent on light intensity, temperature, and abundance of nutrient substances and phytoplanktonic organisms. The rate of decomposition in poorly lit deep waters depends on availability of oxygen, temperature, and abundance of decomposer organisms. All of these processes involve chemical reactions.

According to van't Hoff's Law, the rate of chemical reactions is doubled or tripled by a 10°C increase in temperature. Physiologists have used this law, which they call the Q_{10} rule, to compare the effects of temperature on physiological rates (Hoar, 1975). Ruttner (1931) invoked an ecological Q_{10} rule to explain the differences in limnological features between tropical and temperate lakes as noted above. Regier and Henderson (1973) also referred to an ecological Q_{10} concept with respect to fisheries production in waters of different latitudes. Recent comparative fisheries-related studies of Pauly (1980) and Schlesinger and Regier (1982) implicitly used the Q_{10} rule. George Woodwell (1983) used an ecological Q_{10} approach to derive some first approximations as to the likely impacts of climate warming on terrestrial ecosystems.

The Arrhenius relationship has been used to describe the effect of temperature change on the "specific rate constant," k, in chemical reactions. For many physiological and ecological applications a first-order chemical reaction has often been taken as an analogue of some living process. Here

$$\frac{dx}{dt}\frac{1}{x} = k,$$

where x is the mass or some surrogate measure of living substance. If a particular ecological process can be approximated by a first-order expression, then the relevant Arrhenius expression is

$$k = Ae^{-E/RT},$$

where T is absolute temperature, and A,E, and R are constants. The Arrhenius expression is soundly based in theory and in practice and, therefore, preferable to van't Hoff's empiric expression (Regier et al., 1988). Regier et al. (1988) provide a more detailed rationale for the use of the Arrhenius relationship.

The Arrhenius relationship provides a simple model from which first approximations of some potential changes in lake productivity in response to climate warming can be derived.

METHODS

Data from the literature at the ecosystem level of organization were fitted to the Arrhenius relationship by regression methods, and community Q_{10} values were then estimated for phytoplankton production, zooplankton biomass, and fishery yields. No data were located that treated an ecosystem as an integrated whole, but data sets were found which related three major ecosystem components -- algae, zooplankton, and fish -- to temperature. The Arrhenius expression summarized the information available in the form:

$$\log_e R = a\left(\frac{1}{T}\right) + bV + c$$

where: R is an ecosystem response variable, such as production;

T is absolute temperature,°K;
V is another environmental variable, such as lake depth or fertility; and
a, b, and c are estimated regression coefficients.

Here the Q_{10} measures the proportional change in R with a temperature increase of 10°K.

The multiple regressions relating primary production, zooplankton biomass, and fish yields to temperature (T) and Ryder's morphoedaphic index as V in the above equation are either taken from the literature or computed by us from literature values (Table 2).

To estimate the relation for primary production and temperature in Table 2, primary production and MEI data compiled by Oglesby (1977) and mean annual air temperature data from Wernstedt (1972) were used in a stepwise multiple regression (SAS Version 6.02). Temperature explained 51% of the variation in primary production and MEI explained only 7%, and the model appears to have a good enough fit to estimate Q_{10}.

To estimate the relation for zooplankton biomass and temperature in Table 2, zooplankton biomass, mean epilimnetic temperature, and MEI from Patalas (1975) were used in stepwise multiple regression. Temperature explained 89% of the variation in zooplankton biomass while MEI only explained 3%. The overall model accounted for 92% of the variation in zooplankton biomass among lakes.

The relation for maximum sustained fish yields in Table 2 is a reparameterization of the relationship reported by Schlesinger and Regier (1982).

In addition to the three relationships in Table 2, we also estimated two for marine taxa, one on shrimp with data from Turner (1977), and another for fish stocks with data from Pauly (1980). The temperature coefficients, not shown here, were roughly comparable to those shown in Table 2 which gives us further confidence in our estimates of community Q_{10} values.

RESULTS

Community Q_{10} values ranged from 2.6 for fishery yields to 3.7 for primary production, and 4.0 for zooplankton biomass (Figure 14). When these different processes were fitted by the Arrhenius expression, they provided similar estimates of the temperature coefficients. We do not understand enough about the relevant theory to be able to explain this observation, but from a practical viewpoint we infer that the relevant community rate processes respond, qualitatively and quantitatively, in ways similar to temperature. Temperature effects on ecosystem processes or properties appear greatest on the primary producers and progressively less as one moves up the food web.

Table 2. **Multiple Regressions Relating Community-Level Parameters to Temperature and R. A. Ryder's Morphoedaptic Index, Using the Arrhenius Relationship, and the Resulting Community Q_{10} Values for Phytoplankton Production, Zooplankton Biomass and Freshwater Fish Maximum Sustained Yields[a]**

Phytoplankton Production[b]

(1) $\log_e 1^\circ = 44.40 - 11366(1/T) + 0.2908 \log_e MEI_{25}$

$N = 32$; $R^2 = 0.575$; $r_1^2 = 0.510$; $r_2^2 = 0.065$;
$Q_{10} = 3.7$

Zooplankton Biomass[b]

(2) $\log_e ZB = 40.91 - 12074(1/EPTEMP) + 0.4080 \log_e MEI_{25}$

$N = 14$; $R^2 = 0.918$; $r_1^2 = 0.887$; $r_2^2 = 0.031$;
$Q_{10} = 4.0$

Freshwater Fish Maximum Sustained Yield[c]

(3) $\log_e MSY = 29.74 - 8115.4(1/T) + 0.4895 \log_e MEI_{25}$

$N = 43$; $R^2 = 0.826$; $r_1^2 = 0.737$; $r_2^2 = 0.089$
$Q_{10} = 2.6$

---

[a] Symbols and units: 1° = primary production (g C/m^2.yr); ZB = zooplankton biomass during the summer in wet weight (mg/L); MSY = maximum aggregate sustainable yield (kg/ha.yr); T = mean annual air temperature (°K); EPTEMP = mean epilimnion temperature (°K); MEI_{25} = morphoedaphic index (Ryder, 1982) for maximum mean depth of 25m; Q_{10} = proportional change in ecosystem variable with a 10°K temperature change; N = sample size; R^2 = coefficient of determination; r_1^2, r_2^2 = fractions of variability explained by the first and second variables, respectively.

[b] Derived in this study, see text.

[c] From Schlesinger and Regier (1982).

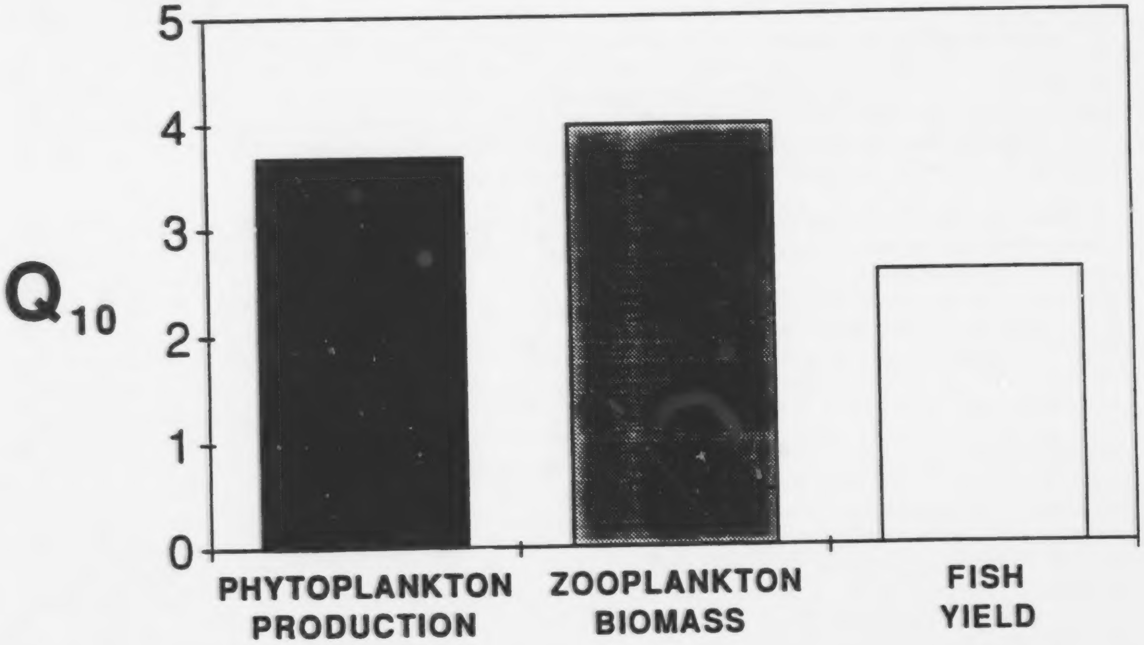

Figure 14. Increases in primary production, zooplankton biomass, and maximum sustained fishery yields expected from estimated community Q_{10} values.

CHAPTER 6

INTERPRETATION OF RESULTS

GENERAL

The initial interpretation of our results suggests that fishes in the Laurentian Great Lakes will benefit from global climate warming as determined from the OSU, GISS, and GFDL $2xCO_2$ climate scenarios. Based on our model simulations, the size of thermal habitat within their optimum temperatures will generally increase in basins with oxygenated hypolimnia (Figures 2, 3, and 4), their growth will increase provided prey consumption increases with climate warming (Figures 9 and 10), population size of an example warmwater fish will increase (Figure 13), and the biological productivity will increase at all levels from from primary production, to secondary production, to fish yields (Figure 14). These results were obtained not only for warmwater fishes like smallmouth and largemouth bass but also for coolwater fishes like yellow perch and walleye, and even for coldwater fishes like trout and salmon. Nor were the increases restricted to the northern cold waters of Lake Superior; similar increases appeared in cool and warm regions including neritic areas of Lakes Michigan, Huron, and Erie.

Our simulations also revealed some potential negative effects of climate warming. During summer, the simulated thermal habitat size for a number of coldwater and coolwater fishes decreased (Figure 5) suggesting that during summer, species interactions such as competition and predation could intensify in the optimum thermal habitats. This restriction in summer habitat could be a limiting factor for some species as it is for certain species in southern U.S. reservoirs (Coutant, 1987b). Simulations of the thermal habitat in brook trout streams during summer showed that suitable temperatures decreased in a similar manner (Figure 7) and could have similar consequences. Increases in fish growth (Figure 9) and its corollary, increased prey consumption (Figure 10), would intensify species interactions and put an increased demand on prey production. In some of the lakes, forage for the salmonid fishery is believed to be at a premium (Stewart et al., 1981), and the increase in demand for forage could result in an overexploitation of prey resources by the salmonids. If additional prey are not available, then the increased temperatures will result in a decline in growth (Figure 11). We have no ecosystem level models for simulating the nature of such surprises.

There are other unknowns for which models or model simulations are unavailable at present. (1) The increase in warmwater habitats may encourage colonization by warmwater exotics (Mandrak, 1988), which could destabilize fish communities in warm areas (Magnuson, 1976). (2) The reduction of ice cover in bays could reduce the reproductive success of lake whitefish (Taylor et al., in press; Freeberg, 1985). (3) If the global climate warming reduced vertical mixing of Lakes Michigan, Huron, and Ontario, and the deeper basin of Lake Erie for a long period of years, the oxygen in these deep lakes could become limiting to the detriment of the coldwater fishes. Our simulations of increased growth assume that the fishes can behaviorally thermoregulate; that is, when inshore or surface waters become too warm, the fish can find suitable thermal habitat offshore or downslope or to the north. This means that the water in and below the thermocline and below must contain sufficient oxygen to support fish life. Presently oxygen is depleted annually in the hypolimnia of the shallower basins of Lake Erie, and these waters cannot support coldwater fishes. The increase in primary production suggested in our simulations could exacerbate this process of developing anoxia in the deep coldwaters by increasing the annual hypolimnetic oxygen deficit. (4) Our analyses have not included any effects on fishes from loss of wetlands caused by water level changes. (5) Our analyses have not included any simulation of the influence of climate warming on the availability and accumulation of microcontaminants in Great Lakes fishes. (6) Our analyses have not included effects of stream warming on the reproductive success of sea lamprey. (7) Our analyses have not included predictions about the smaller inland lakes, which are sure to be affected by climate warming. Our general conclusion about the direct influence of temperature changes on fishes in the waters of the Great Lakes, as induced by global climate change, is that it will be positive. A number of the indirect effects deserve research level attention.

SPECIFIC INTERPRETATIONS

Thermal Niche

Thermal habitats for all thermal groups of fishes (coldwater, coolwater, and warmwater) increase from the BASE climate with the OSU, GISS, and GFDL $2xCO_2$ climate warming scenarios when the entire year is considered. In some cases, such as for coldwater fishes, the increase is dramatic, averaging a factor greater than 2 or a doubling of thermal habitat. The increase results because warmer waters are found in spring, fall, or winter and at deeper depths than for BASE climate. However, during summer, the isotherms are compressed and the size of the thermal habitat (±2°C) in summer decreases under both scenarios by 2 to 47% depending on the species. Inter- and intraspecific competition and predation may be intensified during these months. Fishes may as a consequence occupy a broader range of temperatures than optimum (Rudstam and Magnuson, 1985) and grow less rapidly. The interaction of warmer epilimnetic temperatures coupled with increased hypoxia in the hypolimnia of some Great Lakes basins and bays may lead to a "summer squeeze" on the available habitat space for some species (Colby and Brooke, 1969; Coutant, 1985; Coutant, 1987a,b).

The increased surface water temperature in open water opens up or expands thermal habitat for warmwater fishes in Lake Michigan and in Lake Erie. These conditions would provide more suitable habitat for white bass and white perch, two species which are currently expanding their ranges in Lakes Erie and Ontario. Interest in stocking Lake Michigan with striped bass, which has been introduced to various fresh waters in the southern United States, may increase.

The expansion of optimal habitat of all species in the spring and fall in both climate scenarios suggests that these "shoulder" seasons, if important to yield, may evolve as major contributors to fish yield and production if climate warms. This appears to be the case with another coldwater fish, the burbot, in Lake Opeongo (Hackney, 1973). Of course, this depends on similar changes in other factors that control growth and yield, such as food supply and interspecific competition.

Growth and Food Consumption

The bioenergetics simulations indicate that individual fish will grow faster under $2xCO_2$ climate conditions, if sufficient and appropriate prey are available. If prey consumption does not increase with climate warming, fish growth will decline. If the forage base were to decrease or remain the same following climate warming, predator growth rates would be expected to decrease because of an inability to compensate for the increased metabolic costs of operating at higher temperatures. However, if prey availability increased so that fishes could function at or near their optimal consumption rates, increases in growth rates would be even higher than those reported above.

The Q_{10} estimates of potential changes in zooplankton biomass and fish production indicate that at least part of the forage base may increase with climate warming. This result provides evidence that the assumption of a constant P-value may be reasonable. However, because of the uncertainties in estimating prey availability, the effects of climate change on growth may be over- or underestimated. The dynamics of the Great Lakes food webs (Figure 12) subjected to climate warming must be considered in detail in order to answer the question of whether primary and secondary production will increase enough to meet the increased predatory demands of fishes.

Although the results for consumption are quite similar to those for growth, total consumption is an important parameter in its own right because it is a measure of the trophic interaction among fishes and between fishes and their prey. Increased consumption implies an amplification of competition for food if the forage base does not grow as fast as the predatory demand on it. Consumption may also be thought of as a measure of the potential predation pressure on the forage base. Predation pressure increases significantly for the $2xCO_2$ water temperature scenarios (Figure 10).

Population Response

Smallmouth bass populations in shallow protected bays in warm areas of the Great Lakes would not be harmed and would continue to provide recreational fishing. The fishing would improve in cool areas, and new fisheries would develop in cold areas.

Other warmwater species and warmwater exotics might experience similar changes and opportunities from range expansion.

Ecosystem Response

We infer from our statistical studies that the early observations by Ruttner (1931) and his colleagues were generally reliable. The rates of ecosystemic processes respond to temperature changes in an exponential way broadly consistent with the Q_{10} rule. Most expressions in Table 2 from which Q_{10} values were estimated also include data from the Great Lakes; hence, these broad generalization do apply specifically to the likely effects of climate warming on ecosystemic processes in the Great Lakes.

We expect that climate warming would cause an overall increase in primary production (phytoplankton), secondary production (zooplankton), and maximum sustained yields of fisheries across the Great Lakes (Figure 14). Lower trophic levels would experience a greater increase than higher trophic levels. The increased ecosystemic productivity may be associated with shifts in community structure favoring organisms having higher temperature preferences. The increased biological production will also be reflected in a long-term increase in biogenic materials in the lake basins, which will contribute to increased oxygen demand in the hypolimnia. In basins with small hypolimnia or with permanent thermal stratification, this increased oxygen demand would contribute to hypoxia, with negative consequences for fishes.

CHAPTER 7

IMPLICATION OF RESULTS

The five Great Lakes are inhabited by cold, cool, and warm water fishes and provide a range of thermal habitats characteristic of a large, deep, cold lake (Lake Superior) to a large, shallow, and warmer lake with anoxic hypolimnia in some basins (Lake Erie). The lakes provide important recreational and food fisheries to heavily populated regions of Canada and the United States. For most regions, the climate warming would increase the amount of optimum thermal habitat not only for warm water fishes, but also for cool and cold water fishes. Consequently, the productivity and potential yields from these thermal guilds is predicted to increase in general if climate warming occurs. The productivity of zooplankton and phytoplankton is also expected to increase. New warmwater fish species are expected to invade, especially into the littoral zones of the more southern habitats. Even though productivity is expected to increase in general, surprises involving water quality interactions and loss of important stocks are expected from invasion of new species as well as from intensified species interactions in the fish communities.

CHAPTER 8

POLICY IMPLICATIONS

1. Increased fish production should provide greater recreational, commercial, and artisanal fishery opportunities.

2. Increased value of the Great Lakes in a warmer and dryer time should attract more people to the basin and increase the consumptive use of the Great Lakes' water and living resources.

3. Increased predation pressure on the forage base should require management agencies to pay even closer attention to the management of food webs and prey-predator interactions.

4. The potential for the further invasion of warmwater exotics and expansion of their thermal habitat should require even greater attention to the effects of exotics on the stability and productivity of Great Lakes fisheries.

5. Increasing value of the fishery should continue to make the reduction in toxins that bioaccumulate a high priority for the Great Lakes.

6. Heightened eutrophication effects due to greater primary production and greater isolation of the hypolimnia should require continued and increased attention to nutrient reductions in some lakes.

7. Water for the Great Lakes themselves should be more of an issue for fishery management agencies.

8. Coldwater fishes: thermal habitat for lake trout should increase more than for Pacific salmon; thus rehabilitation of lake trout populations should remain an important policy issue.

9. Coolwater fishes: production of coolwater fisheries such as perch and walleye should increase and should be expected to move northward and offshore.

10. Warmwater fisheries: production of warmwater fishes should increase in cool areas and develop in cold areas. Managing the species structure of the inshore habitat should be an important policy issue.

TERMS, ABBREVIATIONS, AND SYMBOLS

Climate Scenarios

BASE = Scenario using actual air temperatures from 1951-1980.
OSU = Scenario produced by the Oregon State University general circulation model.
GISS = Scenario produced by the Goddard Institute for Space Studies general circulation model.
GFDL = Scenario produced by the Geophysical Fluid Dynamics Laboratory general circulation model.

Areas

Cold = Northern Lake Superior
Cool = Southern Lake Michigan, Baie du Dore (Lake Huron)
Warm = Lake Erie

Fish Thermal Guilds and Temperature Preferences

Coldwater = Lake whitefish, coho salmon (~15 C), lake trout (~10 C)
Coolwater = Yellow perch, walleye (~24 C)
Warmwater = Largemouth bass, smallmouth bass, bluegill (28 C)

Thermal Niche Model

Depth day = The depth of the water column within the thermal niche summed over days
Volume day = The volume of water within the thermal niche summed over days
THV = Summer thermal habitat volume in hm3.7 days or hm3. 10 days (hm = hectometers)
THA = Summer thermal habitat lake bottom area in ha.7 days (ha = hectares)
SY = Sustained yield of fishery in kg/yr

Brook Trout Stream Model

FTw = Change in temperature of parcel of water
Tw = Initial temperature of parcel of water
A = Surface area of parcel of water
V = Volume of parcel of water
v = Volume of inflow (e.g. groundwater, tributary)
tv = Temperature of inflow
S = Change in energy stored in parcel of water

Bioenergetics Model (all units g/day, e.g.)

G = Fish growth rate
C = Rate of food consumption
F = Egestion
U = Excretion
P-value = Proportion of maximum consumption rate

Smallmouth Bass Model

M = Natural mortality rate
Tm = Age of first maturity

TERMS, ABBREVIATIONS, AND SYMBOLS (continued)

F = Fishing mortality rate
YCS = Year class strength
N1 = Year class strength at year 1
NTm = Year class strength at age of first maturity
S = Number of breeding adults
a,B = Empirically determined constants in the equation relating N_1 and S
w = First year winter survival

Q_{10} Model

Q_{10} = Proportional change in R with a 10°K temperature increase
R = Ecosystem response variable
V = Environmental variable
a,b,c = Estimated regression coefficients
MSY = Maximum Sustainable Yield
T = Mean annual air temperature
EPTEMP = Mean epilimnion temperature
MEI25 = Morphedaphic index for maximum mean depth of 25m
1o = Primary production in g C/m2/yr

COMMON AND SCIENTIFIC NAMES OF FISHES CITED IN TEXT

Common Name	Scientific Name
Lake trout	Salvelinus namaycush
Lake whitefish	Coregonus clupeaformis
Coho salmon	Oncorhyncus kisutch
Yellow perch	Perca flavescens
Walleye	Stizostedion vitreum vitreum
Northern pike	Esox lucius
Largemouth bass	Micropterus salmoides
Bluegill	Lepomis macrochirus
Smallmouth bass	Micropterus dolomieui
Sea lamprey	Petromyzon marinus
Cisco (lake herring)	Coregonus artedii
Chubs	various Coregonus sp.
Burbot	Lota lota
Sculpins	family Cottidae
Sauger	Stizostedion canadense
Muskellunge	Esox masquinongy
Lake sturgeon	Acipenser fulvescens
Bullheads	various Ictalurus sp.
Channel catfish	Ictalurus punctatus
White bass	Morone chrysops
Sunfish	family Centrarchidae
Crappies	family Centrarchidae
Minnows	family Cyprinidae
Brook trout	Salvelinus fontinalis
White perch	Morone americana
Striped bass	Morone saxatalis

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ECOLOGICAL EFFECTS OF GLOBAL CLIMATE CHANGE: WETLAND RESOURCES OF SAN FRANCISCO BAY

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FINDINGS[1]

This report examined the effects of global climate change on the deep-water and wetland habitats of the San Francisco Bay estuary. Global warming will result in increased sea level and modification of the freshwater run-off into the estuary.

Three scenarios were used for the sea-level rise rate: a 100-, 200-, and 300-cm rise by the year 2100. The relative rise at any one location in the estuary is a combination of the land surface movement and sea level change. For the San Francisco Bay region, the lowest rate of sea level rise occurs around the Golden Gate, with greater rates of relative sea level rise occurring in the northern and southern extremes of the estuary. Thus, the effects of predicted sea level rise will be magnified by the regional subsidence.

Under the 100-cm rise scenario, sediment entering the bay via the rivers is sufficient to meet the accretionary balance required by the tidal marshes and mudflats, at least until 2040. However, under the 200- and 300-cm rise, the sediment requirement by tidal marshes and intertidal mudflats will exceed the 50% of the annual estuarine sediment input early in the 21st century. If the area of tidal inundation increases owing to levee failure, it is expected that most of the current 100,000 acres of intertidal wetlands (marshes and mudflats) will be converted to subtidal habitats owing to a negative accretionary balance.

The diked and managed wetlands behind levees face a double jeopardy. If the levees are maintained, the diked wetlands will continue to decline in elevation relative to sea level and will require substantial pumping to remove standing water accumulated from land run-off and precipitation. Otherwise the standing water will inundate the marsh vegetation. The cost of levee repair will also increase substantially owing to greater hydrostatic pressure exerted by sea level rise. If the levees fail, over 130,000 acres of diked wetlands (seasonal wetlands, farmed wetlands, and salt ponds) will be converted to subtidal habitats with resulting declines in migratory waterbirds and extinction of many of the region's rare and endangered species. In some regions (such as Suisun Marsh), additional wetlands may be created from higher sea level, but in the south and central bay, intertidal wetland habitat will be completely eliminated owing to measures that will be necessary to protect existing development in low-lying areas.

The hydrologic changes in the estuary owing to the combined effect of rising sea level and changing climate will be significant. Most of the inflow will occur in the winter, and spring flows will decrease owing to smaller snowpacks in the Sierra. The greater volume of water in the estuary will cause salt intrusion into the Delta. To repel this salt from the intakes of the federal and state water projects, additional water (carriage water) would have to be released from upstream reservoirs. Nevertheless, despite increased freshwater inflows, the average salinity within the estuary will increase.

All of the current regression models relating species abundance to flow are based on a positive correlation between flow and abundance. High flows result in regions of high plankton biomass in the San Pablo and Suisun Bays. This region is called the null zone as it is found where the net landward bottom current is zero. The null zone also supports high numbers of benthic invertebrates and juvenile fish. However, under the scenarios projected for sea-level rise, increasing flows will not result in the positioning of the null zone within Suisun Bay, but rather in the western Delta. During two recent drought periods, plankton and fish biomass reached historic lows when the null zone occurred in the western Delta.

The result of increasing salinization of the estuary will be a decline in brackish marsh plant productivity, waterfowl habitat, and estuarine invertebrate populations. In some cases, marine species will invade areas formerly occupied by estuarine species. However, it is anticipated that overall productivity of the estuary will decrease owing to decline in phytoplankton biomass. Perhaps more critical to many anadromous fish and nesting waterfowl will be the increase in salinity in important spawning and breeding areas. These species require nearly freshwater conditions, and the predicted salinity increase will significantly reduce their ability to successfully reproduce.

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Society currently places a high value on both tidal and wetland habitats, investing money to protect, enhance, restore, and manage these areas. Their loss would have significant economic effects and would also threaten the stability of developed areas behind levees. A number of recommendations are made to protect tidal and diked wetland habitats including dredge spoil disposal in diked areas, high tide barriers, and salt barriers. Each of these engineering solutions have significant environmental trade-offs and must be evaluated now before their large-scale implementation.

CHAPTER 1

INTRODUCTION

OBJECTIVE OF REPORT

The objective of this study is to investigate the response of wetland habitats to two attributes of global climate change: general sea level rise and modification of freshwater inflow to the San Francisco Bay estuary.

Sea level change affects the distribution of various wetland types by conversion to deep water habitats due to inundation. While some wetland types are tolerant of long periods of submergence, others such as tidal marshes are severely impacted. The impacts of changing freshwater supply and sea level rise are related -- a reduction in sediment supply from the rivers will increase the rate of tidal wetlands loss.

Freshwater supply to the estuary affects the longitudinal distribution of various wetland types, from freshwater wetlands in the Delta region to saltwater marshes in San Francisco Bay. In addition, freshwater inflow drives the circulation pattern within the estuary and establishes the salinity gradient, which is important in the distribution of the estuary's fish and wildlife populations.

We have used available topographic, photographic, and sedimentation data to assess the ability of Bay marshes to sustain themselves in light of increasing sea level. Sea level rise scenarios were developed for increases of 100, 200, and 300 cm by the year 2100. Sedimentation data and accumulation of sediment within marshes were calculated using a modification of the model given in Hatton et al. (1983). The quantity of sediment required as a proportion of the riverine borne material is then estimated for various water flow scenarios.

To determine the effects of salinization, we compared the abundance or distribution of organisms as related to various flow scenarios analyzed by Williams (1988).

Based on these analyses, we have addressed the potential long-term impacts of global climate change on the wetlands of the San Francisco Bay estuary and the policy implications which must be considered to protect these habitats.

GENERAL DESCRIPTION OF THE ECOLOGICAL SYSTEM

The general features of the San Francisco Bay estuary have been described by Conomos (1979). San Francisco Bay is the largest estuary on the west coast of the United States. The Bay is actually a complex of interconnected embayments, sloughs, marshes, channels, and rivers (Figure 1). It consists of two main components, the northern reach containing San Pablo and Suisun Bays and the southern reach containing San Francisco Bay proper. Ninety percent of the mean annual river inflow to the estuary enters the northern reach through the Sacramento-San Joaquin Delta. Most of the flow occurs in the winter and spring months at which time the estuary functions as a partially mixed system, whereas during summer and fall low-flow periods, the estuary is well mixed.

The estuary has been highly modified by man's activities (Nichols et al., 1986). Water development, in particular, has had tremendous impact on the distribution of plants and animals. Inflow to the estuary is highly controlled by state and federal water projects. The Sacramento-San Joaquin Delta is a critical element in the water management program: freshwater from the Sacramento River flowing through the Delta from the north is diverted to federal and state pumping stations in the southern part of the Delta near Tracy. The integrity of the Delta levees and channels is critical to the pumping efficiency and the maintenance of freshwater flows to the estuary. Nevertheless, at times during low-flow periods, the net flow in the Delta is directed toward the pumping stations instead of seaward through the estuary. Since the inception of the water projects in 1945,

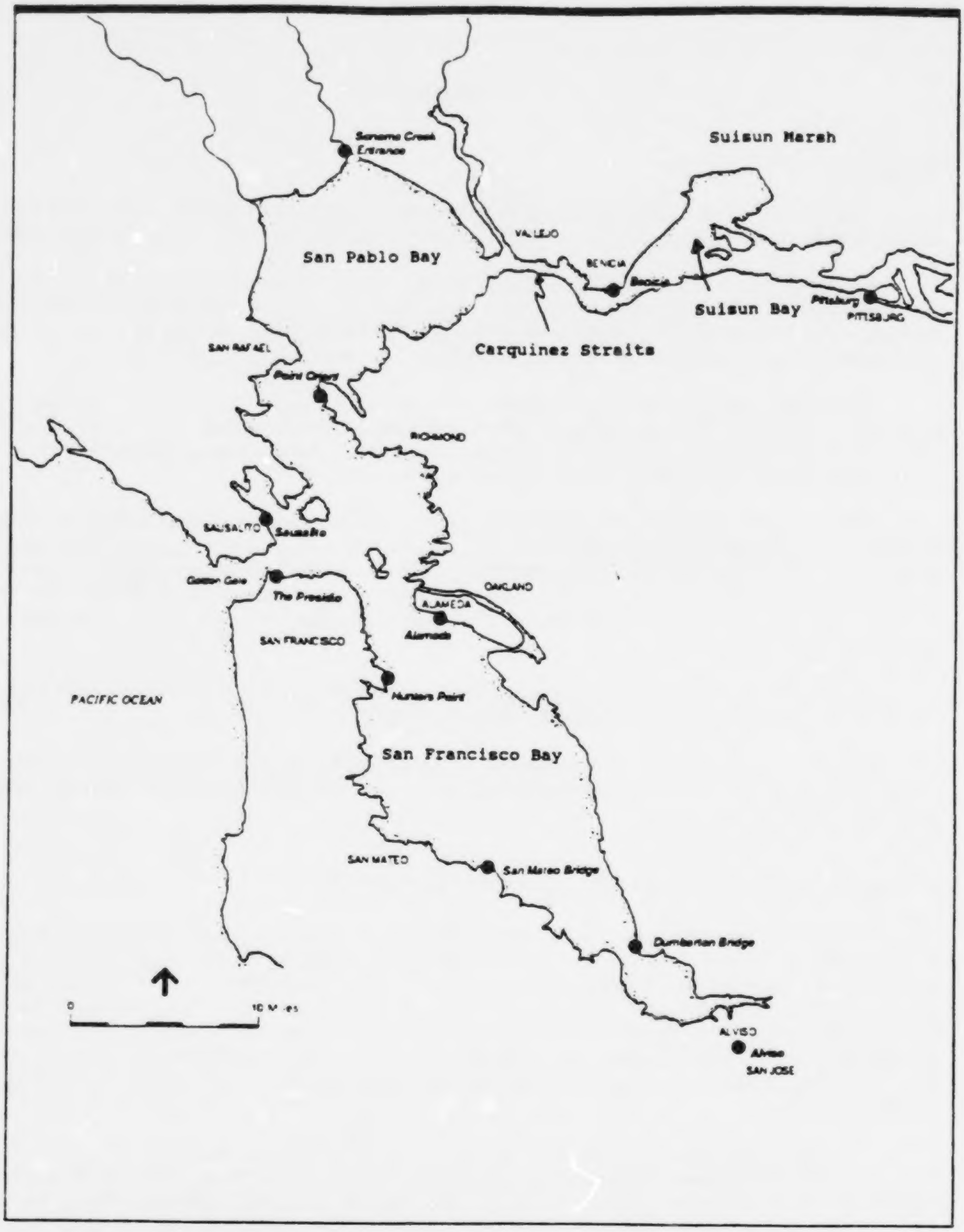
Sonoma Creek Entrance
Suisun Marsh
San Pablo Bay
VALLEJO
BENICIA
Benicia
Suisun Bay
Pittsburg
PITTSBURG
Carquinez Straits
SAN RAFAEL
Point Orient
RICHMOND
SAUSALITO
Sausalito
Golden Gate
The Presidio
OAKLAND
ALAMEDA
Alameda
SAN FRANCISCO
Hunters Point
PACIFIC OCEAN
San Francisco Bay
SAN MATEO
San Mateo Bridge
Dumbarton Bridge
ALVISO
Alviso
SAN JOSE
0
10 Miles

Figure 1. Generalized map of San Francisco Bay estuary with tide station locations (from San Francisco Bay Conservation and Development Commission, 1987).

natural freshwater inflow has been reduced by over 50%, which has resulted in greater salinization of the estuary with concomitant reductions in habitat for anadromous fish and landward shifts in salt marsh vegetation and benthic invertebrates.

In addition to changes in freshwater inflow, other man-induced changes have occurred in the estuarine flora and fauna. Loss of wetlands owing to agriculture and urbanization has resulted in lowered populations of many resident species and in reduced habitat for migratory species. Introduced species of invertebrates and fish have displaced many of the native species. Non-native marsh plants have invaded the remaining wetlands.

Pollution has also affected organisms in the estuary. Domestic waste discharge and agricultural drainage have increased the nitrogen and phosphorus content of the Bay's water. Petroleum by-products have affected the reproductive capacity of striped bass. Heavy metals such as silver and mercury are found in high concentrations in bay invertebrates. Despite all these impacts, however, the San Francisco Bay estuary is relatively low compared to other U.S. estuaries in its susceptibility to pollution and, therefore, still supports important fish and wildlife resources despite significant urbanization (Klein et al., 1988).

The primary focus of this study is on the estuary's wetlands habitats. Atwater et al. (1979) reviewed the history of the tidal marshes around the Bay and their current condition. Josselyn (1983) provided a substantial review of the ecology of tidal wetlands and the San Francisco Bay and Development Commission et al. (1982) of the diked wetlands.

The urbanization of the San Francisco Bay estuary has greatly modified the wetlands around its margins. Recent mapping by the U.S. Fish and Wildlife Service indicates that almost 200,000 acres of various wetland habitats (exclusive of open water) are present, compared to an estimated 540,000 (including the Delta) acres prior to 1850. However, the mixture and type of wetlands present today are primarily the result of human activities such as diking, agriculture, and land development. In most cases, tidal marshes have been replaced by non-tidal wetlands such as salt ponds and seasonal freshwater and brackish marshes. In addition, declines in freshwater outflow (estimated at between 30 and 70% of the natural annual inflow) have resulted in a shift of wetland vegetation from fresh to saline tolerant species. Despite this large-scale conversion and the substantial loss due to urbanization, wetlands continue to provide substantial beneficial uses for flood control, shoreline anchoring, sediment trapping, nutrient retention, and fish and wildlife habitat.

The Bay's wetlands are vital to migratory birds, commercial and recreational fisheries, and a large number of federal and state rare and endangered species. Located along the Pacific flyway, the estuary's mudflats and deep water habitats provide important feeding and roosting areas for migratory waterfowl and shorebirds. Suisun Marsh, the largest brackish water marsh in the western United States, is a major over-wintering habitat for waterfowl and is specifically managed to produce food plants suitable for ducks and geese. The estuary supports several anadromous fisheries including striped bass and salmon. Key spawning areas include the Delta and central valley rivers and streams. Important to the survival of the juvenile fish is the high planktonic biomass occurring in the north bay associated with the null zone or region where fresh and salt water masses meet. As a result of urbanization, a number of plants and animals are now listed as rare and endangered by the federal government. These species, such as the salt marsh harvest mouse, clapper rail, salt marsh bird's beak, and least tern, are largely restricted to marsh habitats. Any further loss or modification of these wetlands will surely reduce these species.

The primary types of wetlands present in the estuary are given in Table 1. The wetland acreages were determined from digitized National Wetland Inventory Maps prepared by the U.S. Fish and Wildlife Service from aerial photography taken in 1985 (Josselyn et al., in press). While over 200 different wetland types were identified, they were grouped into eight major classifications for the purposes of this work: (1) open water (deep and shallow water submerged habitat); (2) mudflat (unvegetated tidal areas); (3) tidal marsh (vegetated tidal areas); (4) diked wetlands (non-tidal wetland areas [exclusive of farmed areas] that are generally within the former boundary of historic tidal wetlands); (5) agricultural lands (former tidal wetlands now used for farming); (6) salt ponds (including crystallizers and bittern ponds); (7) lakes and ponds (shallow open water areas); and (8) other wetland habitats (freshwater and riparian wetlands). Three geographic locations were considered: (1) Suisun Bay between the Delta (at Collinsville) and the Napa River; (2) North Bay between Mare Island and the Oakland-San Francisco Bay Bridge; and (3) South Bay between Treasure Island and San Jose. The areal extent was determined for those areas generally within the 10-ft. elevational contour around the Bay (the Bay Zone).

For Suisun Bay, the largest component is the diked seasonal wetlands, which are managed for waterfowl habitat. The vegetation within these wetlands is typical of brackish water wetlands: alkali bulrush, cattails, brass-buttons, and pickleweed dominate. There is a substantial amount of tidal marsh as well, especially along the southern shoreline of Suisun Bay.

Open water and mudflats are the most abundant wetland types in the north bay, especially along the northern edge of San Pablo Bay. It is also the location of the highest acreage of tidal marsh, much of it created by hydraulic mining debris in the 1860's. Much of the diked historic marshlands are now farmed, although some of the diked areas still function as wetlands. The tidal marshes in this sector are dominated by saline tolerant vegetation: cordgrass and pickleweed.

Table 1. Areal Extent in Acres for Eight Generalized Wetland Types Within the San Francisco Bay Estuary. Based on U.S. Fish and Wildlife Service National Wetland Inventory and Josselyn et al. (in press).

HABITAT	SUISUN BAY	NORTH BAY	SOUTH BAY
Open Water	25,439	101,994	93,220
Mudflats	6,114	28,131	30,378
Tidal Marsh	10,541	16,201	8,604
Seasonal Wetland	47,159	8,606	4,592
Farmed Wetlands	8,263	25,828	1,317
Salt Ponds	27	9,059	27,497
Lakes and Ponds	1,459	2,926	5,331
Other	963	507	1,412
Total	99,965	193,252	172,352

In the south bay, non-vegetated wetland habitats make up the majority of the acreage. Salt ponds in which water depths are controlled provide habitat for a number of fish-eating birds, long-legged shorebirds such as avocets and stilts, and migratory waterfowl. Mudflats provide important habitat for short-legged shorebirds such as sandpipers and willets. Vegetated wetlands and diked wetlands comprise less than 13,000 acres, the smallest acreage of these important habitats for any of the regions in the bay. Pickleweed, a plant tolerant of high soil salinities, is the most prevalent plant.

SENSITIVITY TO PREHISTORIC CLIMATE CHANGE

The evolution of tidal wetlands in San Francisco Bay has been the result of a gradual inundation of lowland areas balanced by sedimentation. The melting of glaciers has been the major cause for a rise in sea level in the past 10,000 years. There have been two phases in the rate of rise during this period (Figure 2). From approximately 10,000 years before present (YBP) to 7,000 YBP, sea level rose at an estimated rate of 20 mm/yr (Atwater et al., 1977). At this rate, dryland was rapidly inundated and converted to tidal flats and

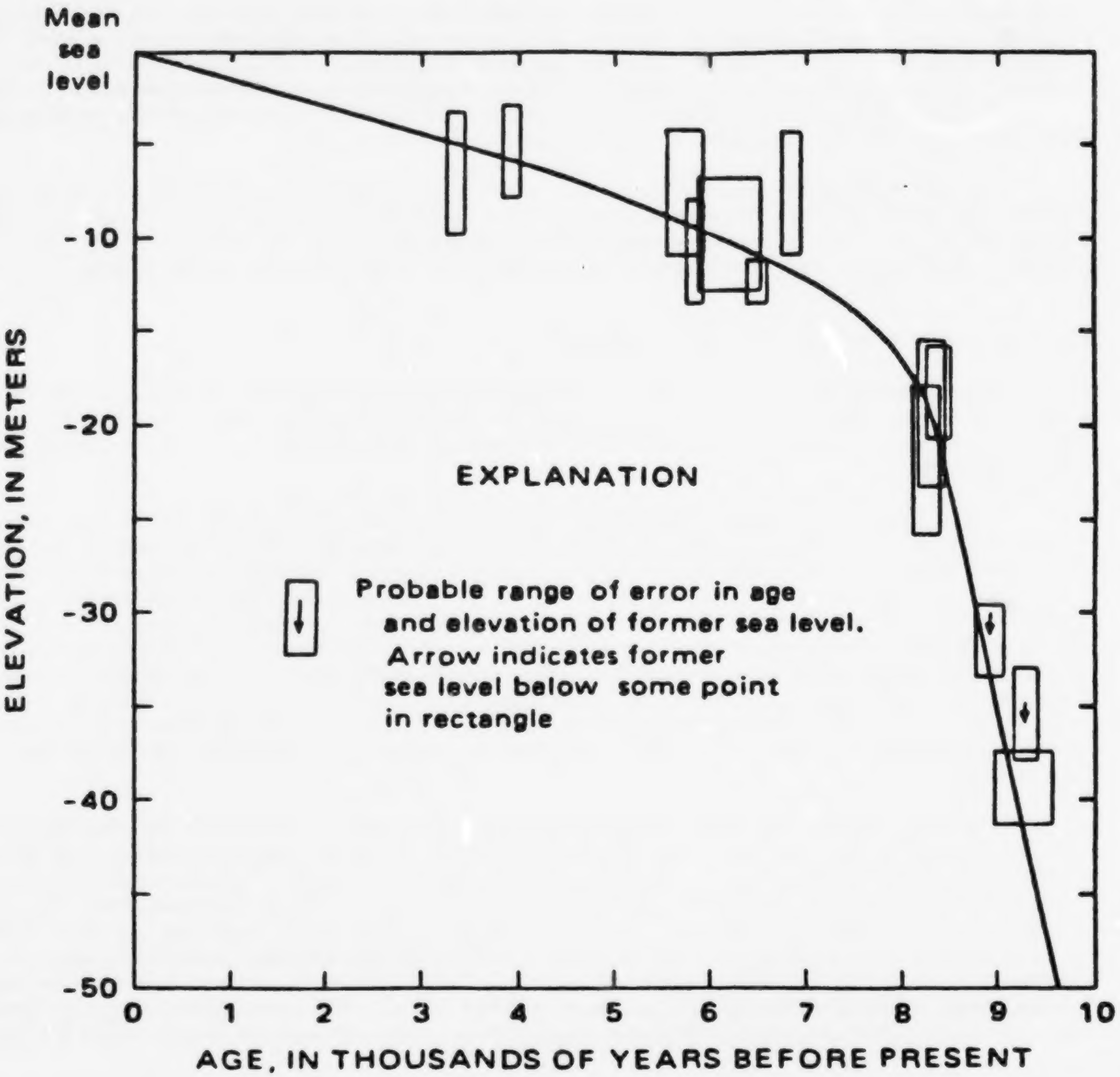

Figure 2. Changes in sea level relative to land during the past 10,000 years at the site of southern San Francisco Bay (after Atwater et al., 1977).

subtidal areas. Dated cores from San Francisco Bay indicate little marsh development during this period. After 7,000 YBP, the rise in sea level slowed to the estimated current rate of 1-2 mm/yr. During this phase of sea level rise, extensive marsh development occurred.

Marsh development along the margins of San Francisco Bay has largely been a function of inundation of dryland by rising sea level. As wetland plants colonized the newly inundated areas, they also retreated from the bay edge as it eroded. Sediments accreted on the marsh surface from this erosion as well as from the suspended sediment derived from river drainage. In the late nineteenth century, increased sediment supply derived from mining and agricultural practices in the watershed resulted in the migration of marshes over former mudflats. With the reduction of these sources and declining freshwater inflow, retreating shorelines along the bay margin are again more prevalent than accreting ones (Atwater et al., 1979).

Current scenarios for global climate change place the rate of sea level rise at between 10 to 40 mm/yr, which is an order of magnitude greater than at present. The assumption is that tidal marshes would be rapidly inundated. However, the acreage of tidal wetland present today is only 10% of that which existed just 150 years ago and, therefore, one might anticipate that sufficient sediment supply is available to support these wetlands.

SENSITIVITY TO PRESENT-DAY VARIABILITY

Two analogues are available to assess the response of present-day wetlands to variability in water flow and sea level rise. The first is the severe drought of 1976-77, which drastically reduced water supply to the estuary, and the second is the relative change in sea level experienced by areas that have subsided due to ground water withdrawal.

During the 1976-77 drought period, water supply to the estuary was reduced by 75% over normal regulated inflow. A number of changes were observed in the vegetative composition of both tidal and non-tidal (managed) wetlands. A substantial decrease in the productivity of brackish water species such as alkali bulrush and tules were observed with a concomitant increase in the occurrence of saline tolerant species such as cordgrass and pickleweed (Atwater et al., 1979; California Department of Fish and Game, 1987). Other estuarine organisms were also affected. Phytoplankton productivity decreased within Suisun Bay (Ball and Arthur, 1979), striped bass juveniles drastically decreased, and more saline-tolerant benthic organisms migrated upstream. These changes occurred within the 2-year period of the drought, and the marine species that invaded the upper reaches of the estuary were eliminated once flows returned to previous levels. However, the recovery of some estuarine species has been much slower. For example, the striped bass population remains critically low to this date.

The second analogue, groundwater withdrawal, has contributed to local subsidence in south San Francisco Bay of up to 1.5 m within the recent past (Atwater et al., 1977). Many diked wetlands formerly at or above MHHW are now at MLLW or below. Continued repair is required to maintain the levees around these areas and pumps are necessary to remove accumulated precipitation. On the other hand, tidal marshes do not appear to have been lost owing to inundation. Patrick and DeLaune (unpublished manuscript, Louisiana State University) used isotopic data to determine sedimentation rates in a tidal marsh near Alviso Sedimentation over a 30-year period averaged 56 mm/yr. At a tidal marsh near Foster City where subsidence is considerably less, sedimentation in a tidal marsh was less than 3 mm/yr during the same time period. Thus, it appears that in some locales, sedimentation can be sufficient to maintain tidal marshes despite large rises in relative sea level.

STUDIES APPLICABLE TO THIS PROBLEM

Most studies provide only a glimpse of the impacts of long-term climatic change on estuarine wetlands and their associated flora and fauna. In general, the focus has been placed on sedimentation rates in wetlands. Stevenson et al. (1986) provide an excellent review of sedimentation studies to date and found a strong positive correlation between tidal range and sedimentation in wetlands experiencing annual sea level increases between

2 to 10 mm/yr. Regions that have a very small tidal range, such as the Mississippi Delta, experience the greatest accretionary deficit as sea level has risen whereas regions with higher tidal ranges have been able to sustain elevations suitable for wetland vegetation.

Regional studies on the impacts of sea-level rise on the wetlands of Delaware and Chesapeake Bays and the Mississippi Delta have also been completed (Hull and Titus, 1986; Stevenson et al., 1985; Salinas, 1986). These studies emphasize the complexity of the factors which contribute to wetland loss in these systems. Salinity intrusion and sediment accretion deficits, however, are the most significant impacts resulting in either modification or loss of wetlands.

The literature for San Francisco Bay is much more limited than that in other areas. Approximations for sediment input to the estuary have been derived (Krone, 1979). The amount of sediment entering the estuary is directly related to total river discharge. In the 1960's, the estimated annual sediment supplied to the Bay via the Delta was 3 million metric tons/yr. By 1990, with projected water development, sediment supply is estimated at only 1.6 million metric tons/yr, a 45 percent decrease.

Less is known about the distribution of sediment once it enters the estuary. Only a few studies have been completed on sedimentation on mudflats or within marshes (Nolan and Fuller, 1986; Pestrong, 1972; Josselyn and Buchholz, 1984). Much of this data is site specific and difficult to generalize to other areas. Mudflats generally exhibit the highest rates of sedimentation, especially in regions where the tidal prism (volume of water between high and low tide) has been reduced owing to filling of wetlands. Accretion rates of 0.6 to 2.8 mm/yr were observed in a shallow embayment, whereas rates of 70 mm/yr occurred in subtidal waters (Nolan and Fuller, 1986).

Within marshes, seasonal variation in elevation may exceed long-term accretion (Josselyn and Buchholz, 1984). Generally, lower elevations within tidal marshes exhibit the greatest rates of sedimentation, which then decrease rapidly with height above sea level (Krone, unpublished). Josselyn and Buchholz (1984) measured annual fluctuation in sedimentation within tidal marsh channels ranging between 100 and 300 mm/yr, whereas Martindale (1987) determined accretion rates of 1-5 mm/yr on the marsh surface.

Recently, a number of studies have been completed by the San Francisco Bay Conservation and Development Commission on the impact of sea level rise on the shoreline of the estuary and the reduction of freshwater flow (San Francisco Bay Conservation and Development Commission, 1985, 1987a,b,c). These reports provide useful recommendations concerning the land-use policy implications of global climate change.

CHAPTER 2

METHODOLOGY

To date, large-scale biological data bases for the estuary have not been integrated into a model suitable to study biological responses to changing water flow or sea level. Therefore, the models used for this study are based largely on empirical data, simple sedimentation budgets, and regression models.

MARSH AND MUDFLAT SEDIMENTATION

Scenarios used for sea level rise were taken from Titus (personal communication). Three scenarios were used: 100-, 200-, and 300-cm rise by the year 2100. The equations used for calculation were:

100 cm scenario: $RSL(t) = L(t) + 0.0012t + 0.000066422$

200 cm scenario: $RSL(t) = L(t) + 0.0012t + 0.00014342$

300 cm scenario: $RSL(t) = L(t) + 0.0012t + 0.00022032$

where: t = the number of years since 1986
$L(t)$ = the local component of relative sea level change
$RSL(t)$ = relative sea level at time t.

$L(t)$ is further refined as $LYLE*t - 0.0012t$ where LYLE is the local relative sea level trend. The value for LYLE for the San Francisco region was obtained from Lyle et al. (1987).

The changes in sea level as predicted under these three scenarios and corrected for the San Francisco Bay region are shown in Figure 3. This model assumes an increasing rate of rise throughout the next century. However, many other models may be equally valid such as a linear rise, a stepped rise, and a precipitous rise.

Within San Francisco Bay, there is considerable variation in local sea level trend owing to differences in crustal subsidence. San Francisco Bay Conservation and Development Commission (1987b) provides local land surface changes for 11 locations throughout the Bay (Figure 4). These data vary considerably, ranging from 0.06 mm/yr in Sausalito near the Golden Gate to 29.23 mm/yr in the south bay at Alviso. To illustrate the range of sea level change that might be expected to occur throughout the estuary, the 200-cm scenario for San Francisco, Sausalito, Pittsburg, and Alviso is plotted in Figure 5. Thus, the actual rise within the estuary for any one scenario may vary significantly depending upon the location. Recently, actions taken to replenish the groundwater aquifers in south San Francisco Bay have halted further subsidence (Santa Clara Water Agency, pers. comm.) and, therefore, the range of potential sea level rise is more appropriately restricted to that shown for Sausalito and Pittsburg on Figure 5.

Sediment budgets for marsh and mudflat surfaces were calculated for each of the three scenarios using the model equation from Stevenson et al. (1986):

Sediment necessary to maintain marsh surface = $BD * PI * AR * A$

where BD = Bulk density of soil
PI = Percent of inorganic matter
AR = Accretion rate
A = Area of tidal marsh or mudflats

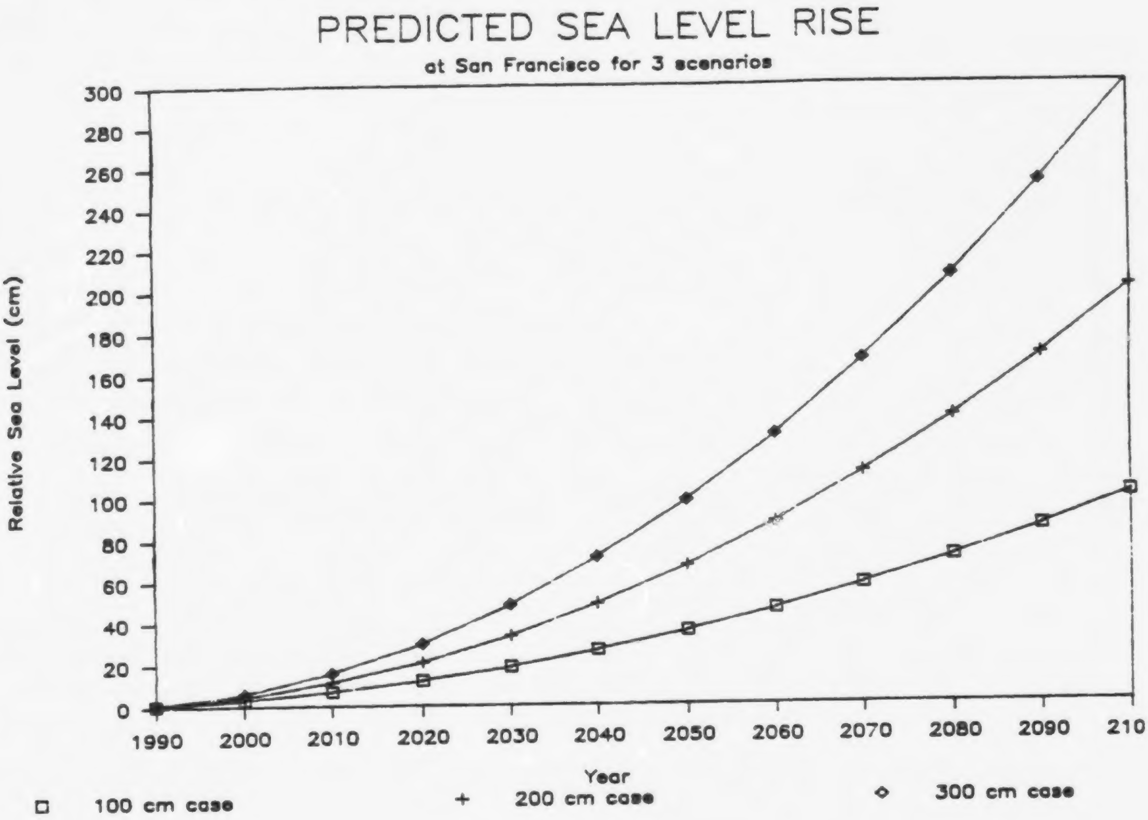

Figure 3. Predicted sea level rise at San Francisco for three scenarios: 100-, 200-, and 300-cm rise by the year 2100.

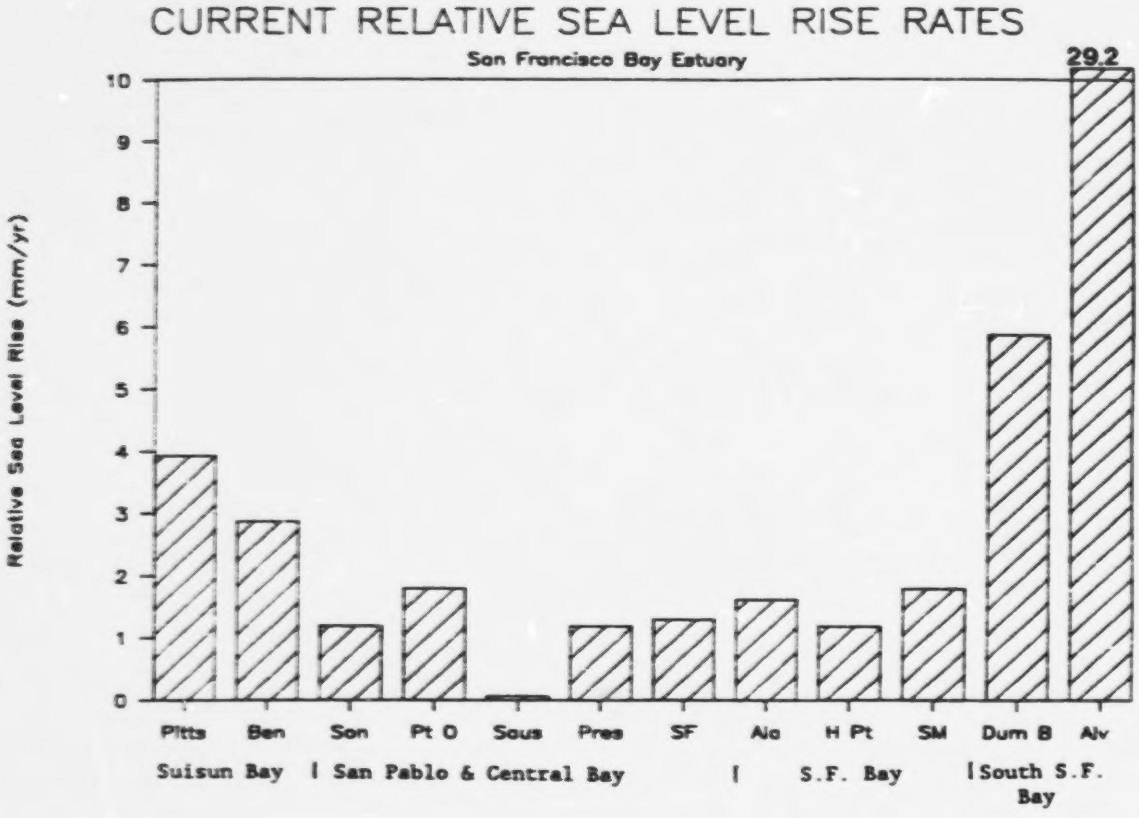

Figure 4. Current relative sea level rise rates for twelve sites in San Francisco Bay estuary. Abbreviations refer to tide station locations mapped on Figure 1 (after San Francisco Bay Conservation and Development Commission, 1987).

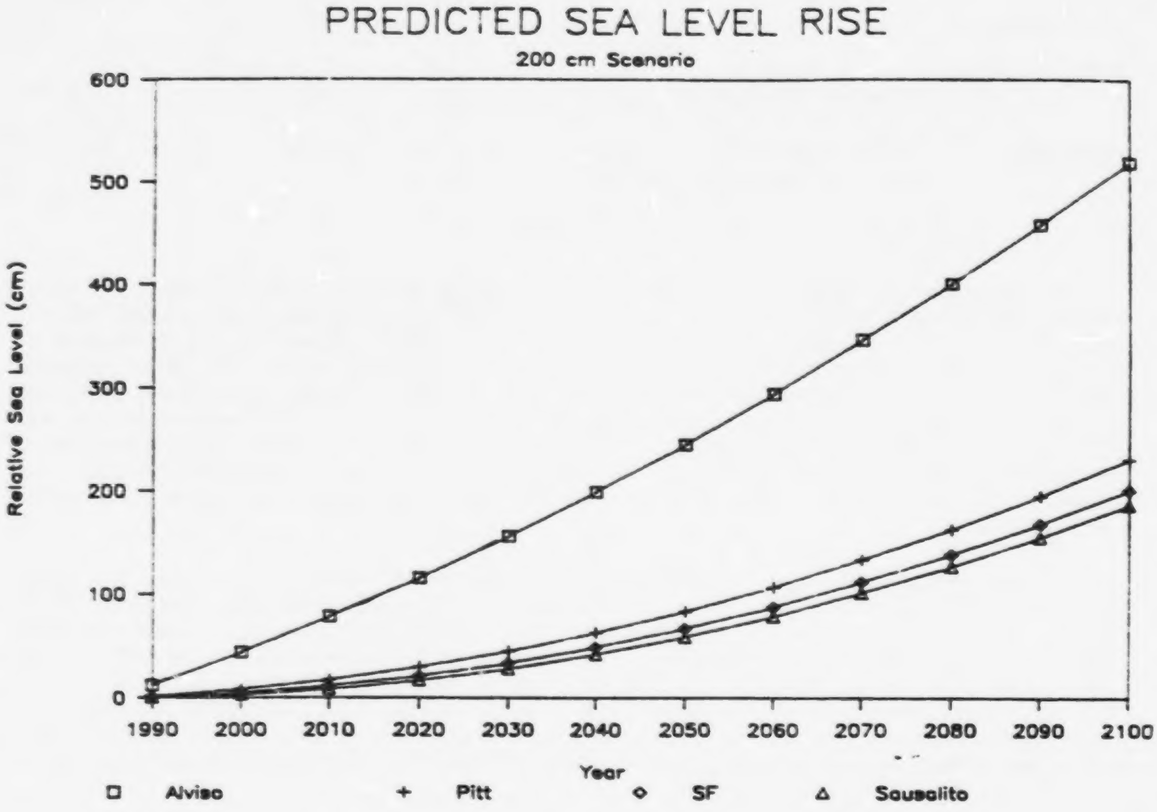

Figure 5. The range of predicted sea level rise in San Francisco Bay estuary by the year 2100, including Alviso Slough, Pittsburg, San Francisco, and Sausalito.

The data for bulk density and percent inorganic material were taken from Pestrong (1972), Mahall and Park (1976), Perez (1981), Nolan and Fuller (1986), and Josselyn (unpublished) for a variety of geographic and wetland types in the estuary. High, median, and low estimates were determined (Table 2).

Table 2. Ranges of Bulk Density and Inorganic Content Used in Determining Sediment Budgets for Tidal Marshes and Mudflats.

WETLAND TYPE	FACTOR	HIGH	MEDIAN	LOW
Tidal marshes	Bulk Density	1.0	0.8	0.65 g/cm3
	Percent inorganic	92	85	78
Mudflats	Bulk Density	1.0	0.8	0.65
	Percent inorganic	98	96.5	95

The sediment quantity required to maintain marsh and mudflat surfaces was then calculated for various accretion rate scenarios within each region of the estuary. Finally, a comparison of the sediment required under each of the scenarios against the sediment available was determined. The annual sediment supplied to the Bay was determined from the relationship of flow to sediment supply as given in Krone (1979). Sediment supply was estimated under four inflow scenarios: BASE, GFDL, GISS, OSU outputs from Williams (1988). An important assumption in Williams's estimate of inflow was that sufficient water would be released by state and federal water projects to retard salt intrusion to water project intakes within the Delta. This additional water is referred to as carriage water. In addition, we assumed that levees around diked wetlands would be maintained. If not, these areas would also serve as sinks for sediment, increasing the percent necessary for marsh accretion.

In addition to sediment budget calculations, marsh erosion was also examined in locations with known historic changes in relative sea level. We initially examined 1:24000 U.S.GS topographic maps from 1945 to present to determine if any areas showed accretion or erosion. The scale of these maps prohibited accurate measurements of shoreline change. Therefore, four locations in the estuary were selected for more detailed analysis of shoreline change: Palo Alto and Alameda Creek Flood Control Channel in the south Bay and Corte Madera and Southampton Bay in the north Bay. Aerial photographs from the 1950's, mid-1960's, and present were enlarged to a scale of 1"=400' and the shorelines overlain. Levees and permanent structures were used for horizontal control. Erosion rates were compared to relative sea level changes for nearby stations.

SALINITY INTRUSION

Changes in water flow and average salinity for the estuary owing to climate change were taken from the studies completed by Williams (1988). He examined the effect of a 100-cm sea level change within several climate model scenarios. With the projected sea level rise, salt water penetrated further landward towards the Delta. In order to maintain water quality standards for water exported from the Delta, greater quantities of freshwater discharge through the Delta would be needed to repel the saltwater intrusion. The estimated "carriage water" required to protect existing water intakes was determined to be twice current levels of discharge. Despite the increased inflow to the estuary, the larger volume of seawater in the Bay resulted in higher average salinities in all of the basins.

Williams also examined the impact of levee failure within the Delta and along the margin of San Francisco Bay. This scenario resulted in a tripling of the existing area of the Bay and a doubling of its volume. The greater volume resulted in stronger tidal currents at restrictions in the estuary (e.g., Golden Gate and Carquinez Straits) and reduced tidal ranges further inland. However, salinity did not increase significantly over the "intact levee" scenario.

A number of linear relationships have been observed between flow and the abundance of certain organisms; however, these regressions are dependent upon the current geomorphology of the estuary. If the Delta islands were flooded, the inland sea created in this region would so alter the geomorphology of the estuary that current regressions would no longer be valid. For example, studies on juvenile salmon migration have shown that higher flows are necessary to assure high survival of smolt by moving them quickly through the Delta region to avoid entrainment losses (Stevens and Miller, 1983). Under the scenario where carriage water is doubled, the regression model shows a higher survival by salmon smolt. However, the hydrology and circulation in the Delta would be entirely different from the conditions on which the regression was based.

Second, most of the biological relationships observed in San Francisco Bay are based on the indirect relationship between flow and salinity. However, under conditions where sea level is rising, increased freshwater inflow does not reduce salt intrusion into the estuary. Thus, existing models of organism response to changing inflow are not applicable and any analysis of the changes which might occur within the estuary under various climate scenarios must necessarily be qualitative.

CHAPTER 3

RESULTS

SEA LEVEL RISE

The rates of sea level rise for 5-year periods under each of the three scenarios is shown in Figure 6. For the purposes of this report, the model was projected for a 50-year period only. The model predicts sea level rate increases throughout the next century. For the 100-cm rise, rates will range from less than 2 mm/yr in 1990 to over 8 mm/yr in 2040. Given the range of accretion rates observed for San Francisco Bay and other wetland systems with similar tidal ranges (Stevenson et al., 1986), it is probable that sediment accretion on the marsh surface could maintain the vegetated wetland for this scenario.

On the other hand, for scenarios of 200- and 300-cm rise by the year 2100, the accretion rates necessary to maintain the marsh surface relative to sea level rise are considerably higher than the average rates presently observed. For the 200-cm scenario, sea level increase would exceed average rates for marsh surface accretion by the year 2005 and, for the 300-cm scenario, by the year 2000. Under the 300-cm scenario, the rate of rise by 2030 will have exceeded that of recent prehistoric rates when it is presumed that tidal marshes were not very prevalent owing to rapid inundation (Atwater, 1979).

The amount of sediment necessary to sustain tidal marshes under various accretion rates is shown in Figure 7. The low, mid, and high estimates are based on the range of data for bulk density and inorganic content given in Table 2. An 11 mm/yr rise would require approximately 1.1 million metric tons (range of 0.8 to 1.5 metric tons) of sediment to be deposited on all tidal marsh surfaces. These estimates are based on present-day tidal marsh acreage. As sea level rises, additional acreage would be flooded. If the levees remained intact, the increase would be negligible. On the other hand, if the levees failed around diked baylands, the potential area over which sediments could be deposited increases by 3 (Williams, 1988). In addition, many of the diked areas are below sea level (compared to tidal marsh surfaces which are generally near MHW) and will be a substantial sink for suspended sediments. Thus, under the assumption of levee failure, the amount of sediment deposited along the margins of the estuary could well exceed 5 million metric tons for an 11 mm/yr rise in sea level.

These data can also be expressed as a percentage of total sediment input from the Sacramento River under various climate model scenarios (Figure 8). The amount of suspended sediments supplied to the estuary were estimated from the regression of flow against suspended sediments given by Krone (1979), multiplied by the flow rates projected by Williams for each of the three climate models. For example, for an 11 mm/yr sea level rise, 20% of the total annual sediment input under the GFDL inflow scenario would be necessary to maintain tidal marsh surfaces relative to sea level rise. However, this is less than the 31% required under base flow conditions because of the greater carriage water flows necessary to repel salt water intrusion.

Under present-day conditions, only 5% of the total annual sediment input is necessary to maintain tidal marshes at equilibrium with sea level rise. Under the lowest projected sea level rise scenario, only approximately 20% of the total sediment input would be needed to sustain marshes (i.e., for the 8 mm/yr rate of rise in year 2040). However, for the highest projected sea level rise scenario, between 35 and 50% of the total sediment input would be needed depending upon the climate scenario used. Again, the amount of additional wetland acreage formed when levees fail will add to the sediment budget requirements.

The preceding discussion did not consider accretion over mudflats. When mudflats do not accrete as rapidly as increases in sea level, erosion of the marsh edge is often the result. Shoreline erosion is a presently a common phenomenon throughout San Francisco Bay (Josselyn, 1983), and we examined the relationship between shoreline erosion and relative sea level change. In all sites examined using aerial photography, significant shoreline erosion had occurred between the early 1950's and the present (Figure 9). Except at Palo Alto, marsh edge erosion rates were generally between 0.6 to 0.9 m/yr since the mid-1950's. Palo Alto exhibited

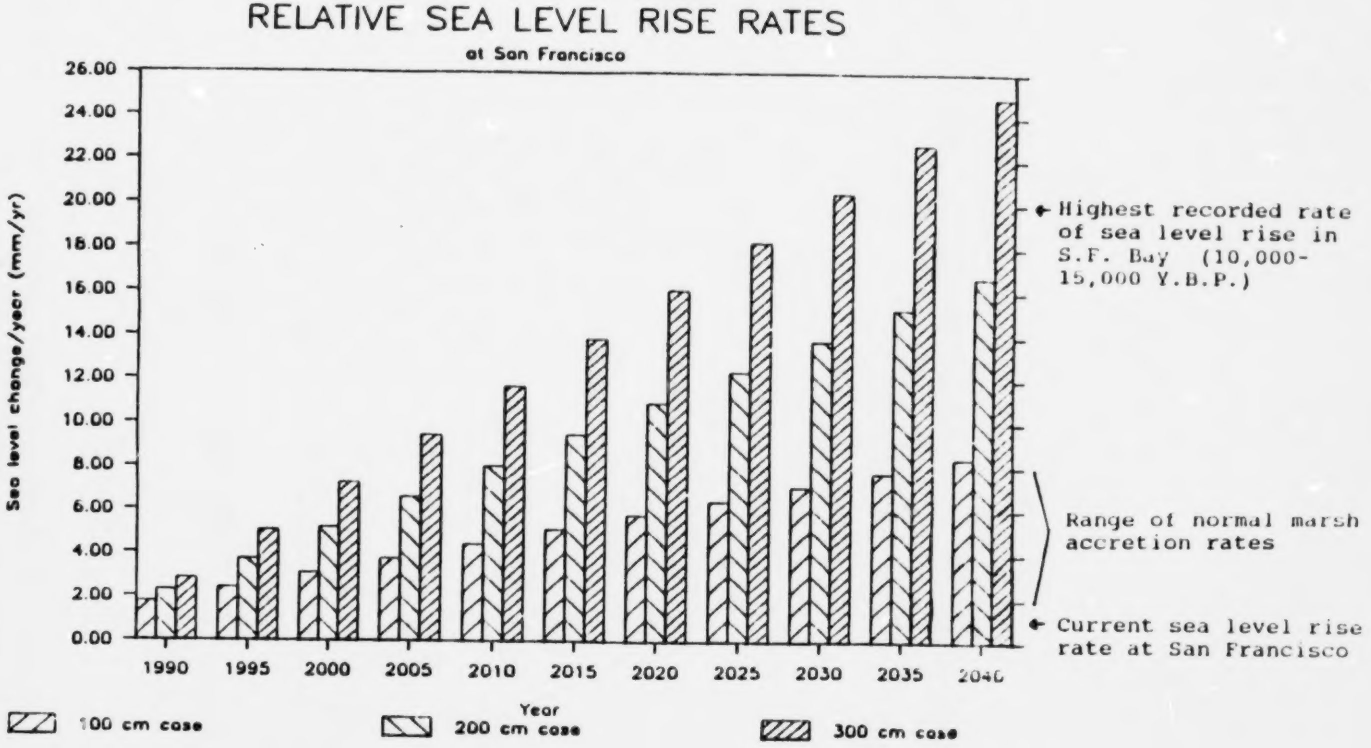

Figure 6. Relative sea level rise rates at San Francisco from 1990 to 2040 under three sea level rise scenarios. Normal accretion rates are taken from sites which have not undergone substantial subsidence.

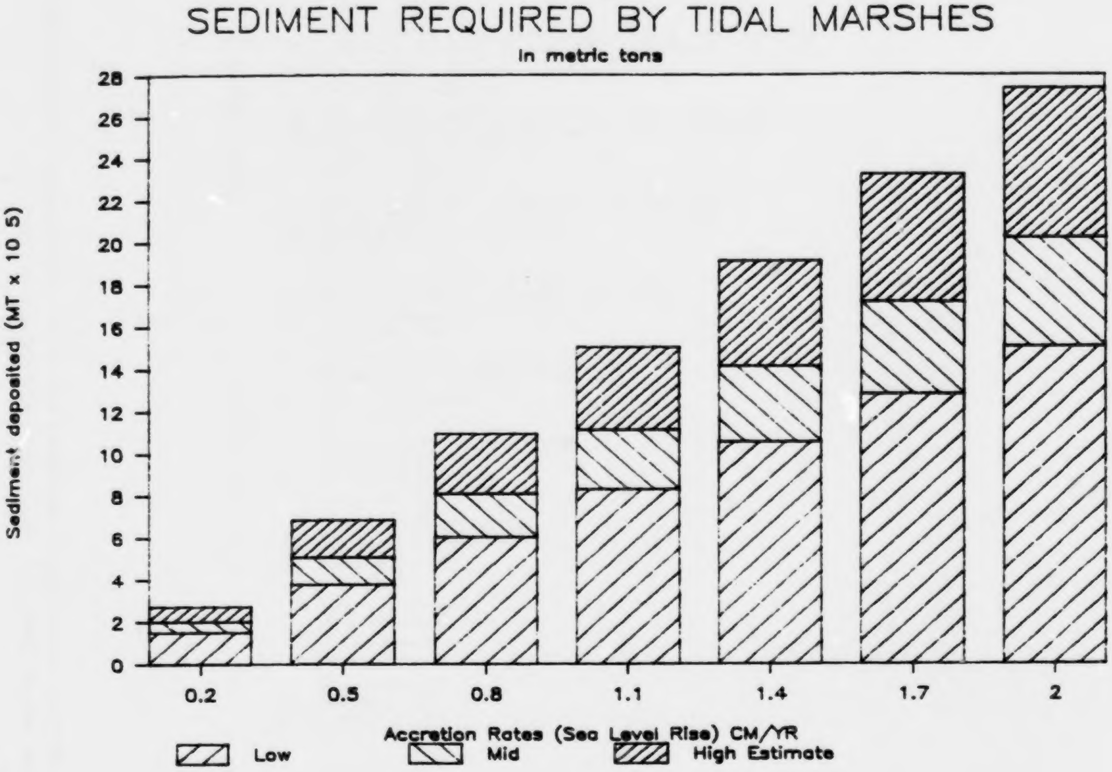

Figure 7. Sediment required by tidal marshes in the San Francisco Bay estuary in metric tons under various hypothetical accretion rates as calculated from Table 2.

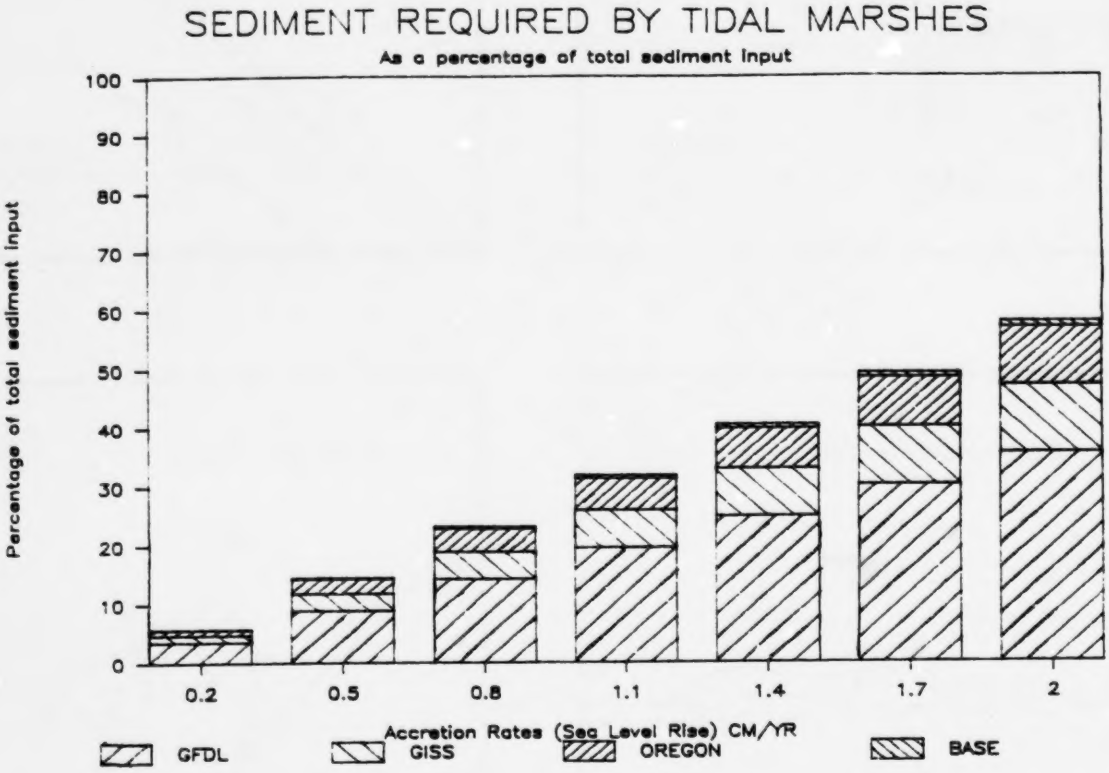

Figure 8. Sediment required by tidal marshes in the San Francisco Bay estuary as a percentage of total sediment input under various rates of predicted sediment input. The percentage of sediment input is indicated by the total bar height for each model scenario. For example, at the 1.1 cm/year rise, the Oregon model predicts 31 percent (the sum of GFDL, GISS, and Oregon bars).

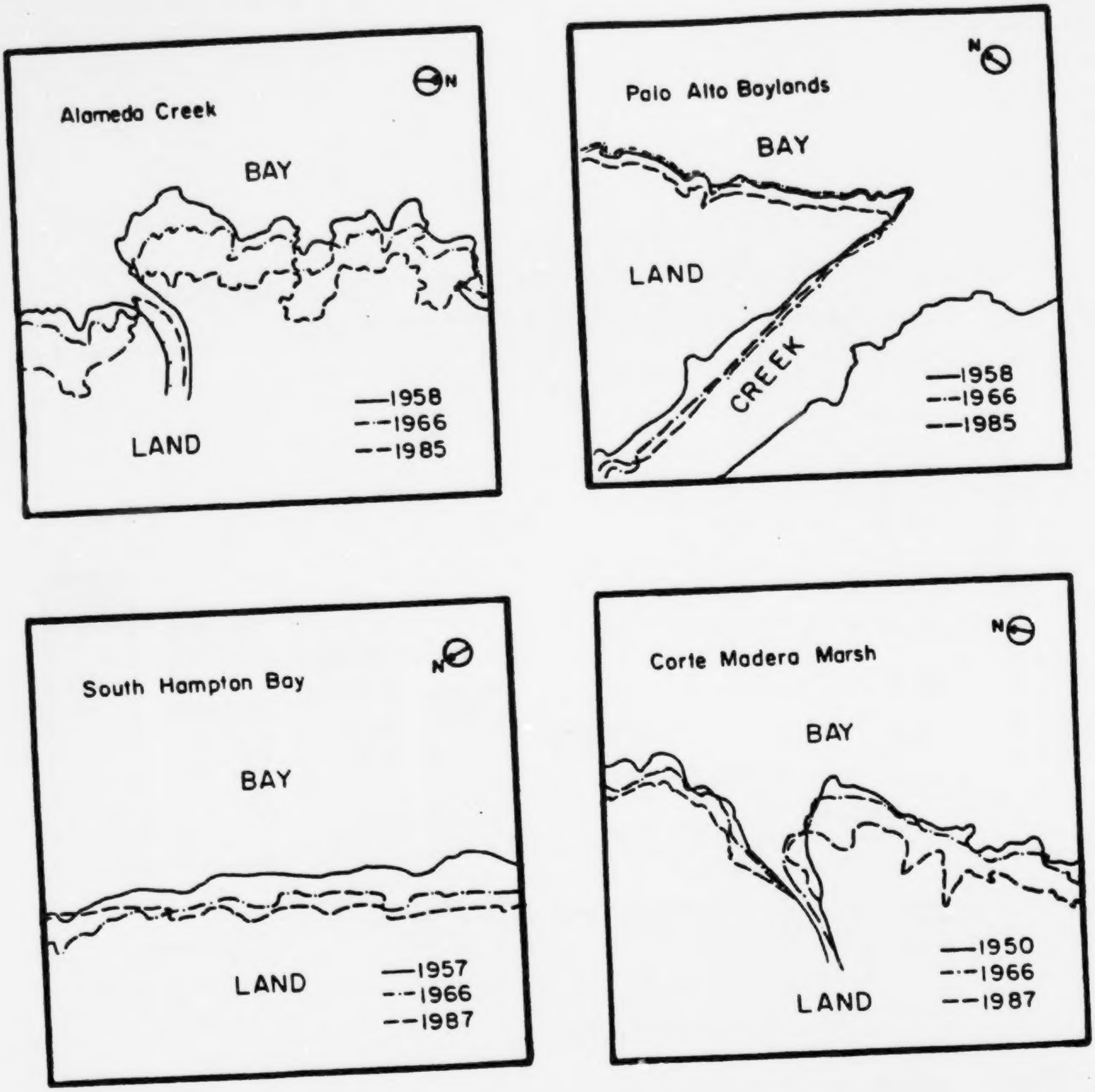
Alameda Creek
N
BAY
LAND
—1958
-·-1966
--1985
Palo Alto Baylands
N
BAY
LAND
CREEK
—1958
-·-1966
--1985
South Hampton Bay
N
BAY
LAND
—1957
-·-1966
--1987
Corte Madera Marsh
N
BAY
LAND
—1950
-·-1966
--1987

Figure 9. Changes in land margins at selected locations in San Francisco Bay.

a rapid rate between 1958-66, but it slowed considerably after this period, partially owing to accretion along the marsh channel. No relationship was observed between relative sea level change and rate of erosion. Thus, given the scale of analysis, a general rate of erosion of 0.75 m is assumed under the most recent past when sea level rise in the Bay area has generally ranged between 2 and 5 mm/yr.

Under the 100-cm scenario, shoreline erosion would probably continue at the present rate up to 2040. However, both the 200- and 300-cm rise scenarios double the probable erosion rate by 2030 and 2010, respectively. This would have serious consequences for the narrow fringing tidal marshes that surround much of San Francisco Bay, especially as many are outboard of levees that do not allow for marsh advance landward.

FLOW MODIFICATION AND SALINITY INTRUSION

As explained above, it is much more difficult to predict the the changes in estuarine communities resulting from modifications of flow or salinity. The detailed modeling does not exist and one must rely on regression analyses. The results discussed below focus on marsh plant distribution, rare and endangered species associated with marshes, phytoplankton biomass, invertebrate distribution, and fish abundance.

Marsh Plant Distribution

The longitudinal distribution of wetland plants within San Francisco Bay is directly related to the salinity gradient. Suisun Bay is considered the transitional zone between the freshwater wetlands of the Delta and the salt marshes of San Pablo and San Francisco Bays. Spring salinities are most crucial to the development of wetland vegetation. Using the 1976-77 drought analogue model for wetland plant distribution, the spring salinities predicted for Suisun Bay with the OSU hydrology and levee failure would reduce the productivity of brackish and freshwater species such as cattails, tules, and alkali bulrush and favor more salt tolerant species such as pickleweed and saltgrass. Salinities predicted under the GFDL and GISS models are within the acceptable range for brackish water vegetation.

The submergence tolerance of many brackish water species decreases as salinity increases (Atwater et al., 1979). Thus, they cannot extend as far down the intertidal. The net effect is a restriction in the intertidal range of species such as hardstem tules, Scirpus spp., the major shoreline plant in Suisun Bay. This shift will reduce available habitat, primary production, and shoreline stabilization.

The impact of the changes predicted under the OSU model would be a reduction in waterfowl use of Suisun Marsh. Approximately 3 to 5% of the state's migratory waterfowl utilize Suisun Marsh during the winter months, feeding primarily on fresh and brackish water vegetation. If the levees around the managed wetlands of Suisun Marsh failed, the existing elevations would not support emergent vegetation and, thus, waterfowl would have to seek wintering grounds in the Central Valley. In addition, waterfowl nesting in Suisun Marsh would decline owing to higher summer salinities projected under all scenarios.

Rare and Endangered Species Associated With Wetlands

A number of federal and state rare and endangered species are associated exclusively with the salt and brackish wetlands of San Francisco Bay. These species include the salt marsh harvest mouse, the California clapper rail, the least tern, Suisun shrew, and the black rail. In addition, other species which may be considered candidates for listing such as the salt marsh song sparrow and the yellowthroat are found within vegetated wetlands.

These species are generally restricted to small home ranges within vegetated wetlands. Other migratory species such as the least tern and snowy plover nest in low-lying areas adjacent to wetlands. Loss of vegetated wetlands owing to either gradual inundation by sea level rise or to catastrophic flooding following levee failure will cause local extinction of these species. As distances between the remaining populations increase, the chances for long-term survival of the species are greatly reduced.

Phytoplankton Biomass

There are two peaks of phytoplankton biomass in the San Francisco Bay estuary: a early spring peak in San Pablo Bay and a summer maximum in Suisun Bay. Generally, the peaks are associated with a circulation phenomena called the null zone: the region where net bottom flow is zero. The null zone has longer residence time than other areas of the estuary and, therefore, phytoplankton biomass accumulates. Flows necessary to sustain these peaks are approximately 20,000 cfs during April and May for San Pablo Bay and at least 15,000 cfs during June and July for Suisun Bay (Williams and Hollibaugh, 1987a,b). These flows generally result in average water column salinities between 2 and 6 ppt in these basins during the period of peak biomass.

The projected salinities under the OSU hydrology indicate that the null zone and, hence, the phytoplankton biomass maxima would be reduced substantially in the San Pablo and Suisun Bays. Spring and summer peaks would, instead, shift upstream into the Delta. However, under GFDL and GISS conditions, it is possible that biomass maxima could still occur depending upon whether or not the levees are maintained. Levee failure would introduce vast expanses of shallow water and mudflats over which primary production may be enhanced.

During the 1976-77 drought and the 1987-88 dry period, phytoplankton biomass in Suisun Bay was near zero during the summer months. Zooplankton, especially a mysid shrimp, Neomysis mercedis and a copepod, Eurytemora sp., were also low. These two zooplankters are the major food source for juvenile striped bass, Morone saxatilis. As a result, survival of juvenile striped bass during both dry periods was at historic lows. Based on these two analogues, we conclude that positioning of the null zone landward of Suisun Bay will result in reduced primary and secondary productivity with a direct effect on an economically important fishery.

Benthic Invertebrates

Many of the invertebrates now dominant in San Francisco Bay are introduced, non-native species. They are considered estuarine, opportunists in that they are tolerant of relatively low salinities and do not occur at all in the marine environment outside the Bay.

One explanation given for a decline in phytoplankton biomass during drought years is the invasion of benthic marine filter feeders into San Pablo and Suisun Bays (Thompson and Nichols, 1987). These species are very effective in removing phytoplankton, and it was estimated that they are capable, in the densities observed, of filtering the entire volume of the estuary in a single day.

In addition to increasing abundance of marine infauna, epifauna such as dungeness crab also increased during the periods of higher bay salinities.

Fish Abundance

Two sets of fish data exist: commercial catches of species such as salmon, striped bass, and American shad and shorter-term data on fish populations taken under the State Board's Bay-Delta program. Most of the commercial data is for the pre-water project development period. Only salmon have a long-term record comparable to the base period selected for this study (1950-81). Rozengurt et al., (1988) examined a number of flow/catch relationships and determined that the best correlations occurred with lag periods of 4 to 5 years. That is, the flows occurring in the previous years had the best relationship to the catch in any given year. The lag time is related to the age at which the fish reached spawning maturity. Two significant direct relationships were observed: the preceding year(s) annual flow with total fish catch and preceding year(s) spring flow (Apr, May, June) with total fish catch. Though the specific regression equations differed among species, the relationships were similar for salmon, striped bass, and shad.

When examining the flow scenarios produced by the hydrologic models, all scenarios provided in Williams (1988) require higher total annual flow to reduce salt intrusion into the Delta due to sea level rise (Figure 10). These annual flows are significantly greater than occurring presently and the regression using annual flow predicts greater fish abundance. However, the Williams report also predicts lower flows in the spring months (April through June). High spring flows have been shown to be the most critical to young fish survival and to greater abundance of spawning fish in subsequent years. Rozengurt et al. (1987) and Kjeldson et al. (1981) recommend spring flows greater than 40,000 cfs to support anadromous fish. None of the climate scenarios provide sufficient flows to meet these criteria, and these fish populations would likely decline.

A complicating factor to these scenarios is the probably of levee failure in the Delta creating a large inland lake with fresh to brackish water quality. Despite the larger winter flows predicted by Williams, average salinities in the western Delta are predicted to increase over base conditions. Striped bass and shad spawn in essentially fresh water conditions, which under base conditions occur during winter and spring months in the Delta. The predicted increase in salinity would eliminate this region as a spawning ground.

A short-term data base (1979-1987) for non-commercial fish species illustrates the difficulty of assessing long-term changes in the estuary's fauna (California Department of Fish and Game, 1987). This report examined the relationship between inflow and monthly fish abundances at over 70 deep and shallow-water sampling sites in San Francisco Bay. High correlations between flows and abundance were observed for a number of crustaceans and fish. For these organisms, winter and spring flows (February-May) were commonly correlated with abundance. However, for a great many species, no correlation or negative correlations were observed. In addition, distribution in relation to salinity varied between years for many of the species and, therefore, no reliable regression with salinity could be developed.

Comparing the flow and salinity scenarios developed by Williams with the Fish and Game data base indicates that despite increasing flows, the higher salinities predicted for the Suisun and San Pablo Basin will increase the abundance of marine species occurring within these basins while decreasing freshwater and anadromous species. Specifically, species such as California halibut, jacksmelt, and walleye surfperch will increase their use of the estuary, while species such as starry flounder, threespine stickleback, and longfin smelt will decrease. The data are insufficient to determine whether this species shift will result in lower or higher fish biomass within the estuary.

SUMMARY OF RESULTS

Specific responses of estuarine organisms to global climate change are difficult to assess from current models. Presently, increased flows result in decreased salinities within the estuary and the available regressions between flow and plant or animal distribution are influenced by this correlation. Under the scenarios investigated by Williams, the necessity to increase carriage water flow to retard salt intrusion to the state's water supply does not result in decreased salinities in the estuary. Instead, average salinity in the basins also increases owing to the larger volume of water present. The resulting conditions are not within the bounds of current regression models. Therefore, our conclusions are primarily qualitative (Table 3).

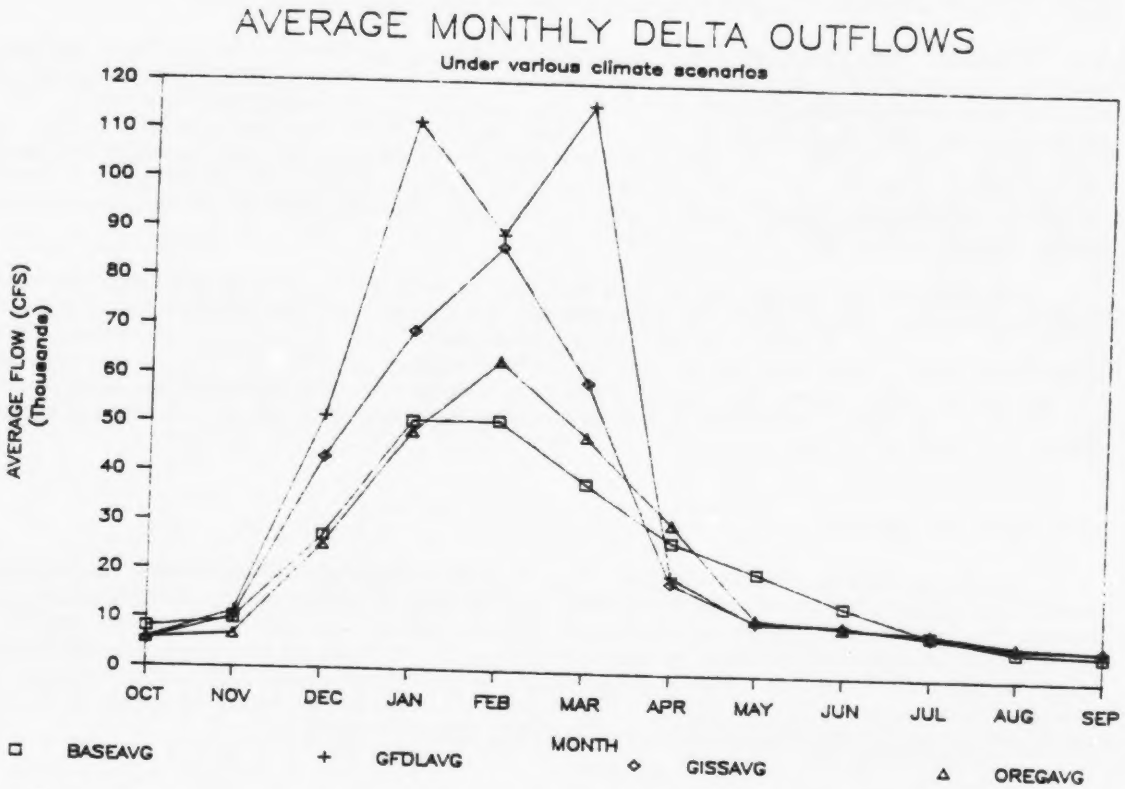

Figure 10. Projected monthly Delta outflows under various climate scenarios (from Williams).

Table 3. Summary of conclusions on the impact of global climate change on estuarine habitats and organisms

HABITAT OR ORGANISM(S)	SCENARIO	LIKELY RESPONSE OR CHANGE DURING PERIOD 1990-2040
Tidal marsh	100 cm rise	Some decline in acreage, but marshes able to maintain accretionary balance; landward shift in distribution of plant species; decrease in production of freshwater/brackish water plants.
	200/300 cm rise	Likely inundation of many of the fringing tidal marshes by end of period.
Diked baylands	All sea level rise scenarios and with levee maintenance	Increasing degradation of marsh vegetation and water quality due to inability to circulate water within diked wetland channels, decline in brackish water vegetation used by wintering and nesting migratory waterfowl.
	With levee failure	Conversion to open water habitat, loss of waterfowl nesting habitat.
Rare and endangered species	All models	Decreased wetland habitat due to tidal marsh loss from increased shoreline erosion, localized and regional extinction.
Phytoplankton biomass	GISS, GFDL with levee failure	Some reduction due to shift upstream during some years
	OSU with levee failure	Shift of peak biomass upstream in all years, decline in zooplankton biomass and lower survivorship of juvenile striped bass.
Benthic invertebrates	All models	Invasion of marine species and displacement of existing estuarine species.

Table 3 (continued)

HABITAT OR ORGANISM(S)	SCENARIO	LIKELY RESPONSE OR CHANGE DURING PERIOD 1990-2040
Fish distribution	All models	Shift upstream in distribution, greater abundance of marine species.
Anadromous fish	All models	Decline in spawning area within delta due to higher salinities; likely decline in abundance due to lower spring flows.

CHAPTER 4

IMPLICATIONS OF RESULTS

ENVIRONMENTAL IMPLICATIONS

Under the moderate and highest sea level rise scenarios, tidal marshes will be inundated as sediment supply will be insufficient to maintain an accretionary balance. This will result in significant losses of animals and plants associated with tidal marshes and adjacent mudflats. Migratory waterfowl and shorebird habitat will be completely inundated. A number of rare and endangered species will become regionally extinct.

On the other hand, the implications to the estuary as a whole may not be significant. This is largely due to the substantial historical losses of tidal marshes which have already occurred. At present, only about 10% of the historical tidal marsh acreage surrounding San Francisco Bay exists. In the past, the area of marshland around the bay was twice as large as the open water area; ow it is less than 10%. Unlike the vast tidal marshes of the southeastern United States that export primary production to the estuary, phytoplankton are responsible for the majority of primary production within the San Francisco Bay estuary (Spiker and Schemel, 1979).

Levee failure will allow transgression of the Bay over former tidal marshes. However, it is unlikely that new emergent wetlands will be created for several reasons. First, most of the diked baylands are currently at or below sea level at present. In the Delta, some lands are 20 feet or more below sea level. Given the sediment deficit predicted, these lands may not accrete rapidly enough to create emergent wetlands. Second, many of the low-lying areas at elevations which might support emergent wetlands upon inundation are currently urbanized.

If levees are maintained, the diked wetlands behind these levees will degrade owing to the inability to flush water through the channels without the use of large pumping systems. These areas will pond water during the winter and then gradually dry out during the summer with a concomitant build-up of salts in the soils. Presently, salt build-up in low-lying diked wetlands of Suisun Marsh and in south San Francisco Bay is causing a die-back in salt marsh vegetation and anoxic conditions within marsh channels.

Regardless of whether or not levees are maintained, the primary faunal impact will be the loss of vegetated tidal marsh habitat that supports rare and endangered species.

The changes in flow and salinity due to modification of freshwater inflow are more difficult to assess. Marine and estuarine organisms will be distributed more landward due to the increases in average salinity throughout the basin. Circulation of the estuary will change dramatically as sea level rises and if levees fail. Changes in sedimentation, retention times, and gravitational circulation are unknown, yet are crucial to understanding the biological responses. The predicted reductions in spring and early summer flow are the most important to the functioning the estuary in its current configuration, especially to the transport of juvenile anadromous fish such as salmon and striped bass.

SOCIOECONOMIC IMPLICATIONS

Extensive public and private resources are invested in tidal and managed wetlands. State and federal agencies require creation or restoration of wetlands as mitigation for development in wetlands. To date, over 50 projects involving wetland creation have occurred in the estuary, ranging from 1 to 500 acres. In addition, the 44,000 acres of managed wetlands in Suisun Marsh represent a significant investment by public and private interests. These wetlands are used primarily for recreational hunting and fishing, itself an industry valued at over $150 million annually (Meyer, 1987). The state and federal governments are investing over $60 million dollars to construct new water supply facilities to the managed wetlands to reduce current salinity problems.

There are also indirect values attributable to public use of wetlands such as tourism, day recreation, and non-consumptive uses. Finally, non-marketable values such as rare and endangered species and heritage values must be considered.

The losses of wetlands, especially diked wetlands if levee failure occurs, would result in substantial economic losses for the region. If levees are to be maintained, the costs would also be significant. The State of California is currently embarking on a 10-year, $100 million levee stabilization project for the Delta. If the higher ranges of sea level rise prove to be true, the cost of levee repair may be insurmountable.

CHAPTER 5

POLICY IMPLICATIONS

San Francisco Bay conservation and Development Commission (1987b) has reviewed some of the local policy decisions which must be made to deal with projected impacts to the estuarine shoreline. Generally, these deal with implementing designs that take into account projected increases in sea level. The report also suggests the establishment of monitoring stations around the bay to determine long-term changes in sea level and provide data to estimate future changes.

The most significant impact to the estuary will be the decision to maintain or abandon the current levee system. There is no choice for areas which are currently developed such as Foster City and Corte Madera. Levees will require replacement and pumping systems will have to be constructed. The levees surrounding less valuable land (agriculture and diked baylands) may not receive the same priority. Many of the levees are poorly engineering (if engineered at all) and are subject to increasing rates of failure due to high tides and floods. Placing more rip-rap or higher levels of fill on these levees will not be an acceptable solution. However, maintenance of the levee system will be essential to reduce the amount of carriage water necessary to repel salt water intrusion and to reduce flooding hazards to developed areas behind the diked baylands.

Several engineering alternatives are possible. One is the construction of high tide/flood barriers which will reduce the impact of high tides. Similar barriers are under construction on the Thames River and the Venice Lagoon. The cost would be substantial for San Francisco Bay and would have significant economic, environmental, and aesthetic impacts. However, they would provide protection to both developed and undeveloped areas.

A second alternative involves the construction of a salinity barrier to retard landward movement of salt as sea level increases. The barrier could be a subsurface sill, a perforated wall, or a constricted channel located along the Carquinez Straits. Whatever design is used, its function would be to regulate the location of the null zone. Although there are a number of environmental problems associated with such a barrier, its primary benefit would be a reduction in the amount of carriage water necessary to repel salt intrusion into the Delta.

A third engineering alternative is to build up the elevations of diked baylands using dredge spoil or other fill material. This alternative would require making decisions which have severe short-term impacts such as the temporary loss of seasonal wetlands (some of which harbor endangered species) in order to manage for long-term impacts of inundation if the levees fail. Given the critical shortage of both tidal and seasonal wetlands in the bay, few agencies will have the ability to suggest any alteration. Nevertheless, it is important that some initial projects be started to demonstrate the feasibility of altering elevations in diked wetlands with subsequent restoration to functioning wetlands. Otherwise, the ultimate fate of these areas will be complete loss.

ADDITIONAL RESEARCH

All phases of estuarine research need to be continued, especially in the area of circulation modeling and factors affecting the distribution of organisms in the estuary. However, one area which is most lacking is an understanding of sediment dynamics in the San Francisco Bay estuary. Sedimentation and sediment movement will play a crucial role in whether marshes, mudflats, and diked baylands will be sustained under projected climate scenarios. Most of the models are based on relatively low rates of marsh accretion determined under current sea level rise. However, several studies reviewed in this report showed that in areas of subsidence, sedimentation can be extremely rapid. Very few measurements of marsh accretion have been conducted in such conditions. In addition, the rate of input of sediment relative to freshwater inflow needs to be determined.

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PROJECTED CHANGES IN ESTUARINE CONDITIONS BASED ON MODELS OF LONG-TERM ATMOSPHERIC ALTERATION

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ACKNOWLEDGMENTS

This report is based on the work of literally hundreds of people over an 18-year period of study. The funding of this work has been diverse, with various agencies (local, state, and federal) contributing over the years to the final product. The Apalachicola data base is the product of the cumulative efforts of students (undergraduate and graduate), faculty, independent researchers, staff personnel, and a long line of volunteers who have given their time in a most generous fashion. A series of project managers and patrons have allowed the continuous accumulation of field and experimental data during this period with the result that we are now in the final stages of analysis, modeling, and publication of the results. Appendices cited in the text of this paper were not included in this volume because of space considerations; however, they may be obtained from Jim Titus, OPPE, EPA. The appendices are:

I. Analysis of existing (long-term, interdisciplinary) field data concerning a series of river-estuaries in the southeastern United States (South Carolina, Georgia, Florida, Mississippi, Alabama).

II. Curriculum vitae: Robert J. Livingston.

III. Statistical analyses of epibenthic macroinvertebrates of the Apalachicola estuary.

IV. Statistical analyses of fishes of the Apalachicola estuary.

V. Data concerning sea level changes in the Apalachicola region. (Dr. Richard Park, Butler University)

VI. Use of existing field data to evaluate the effects of projected long-term climatological changes in a south temperate gulf estuary. (R.J. Livingston, Florida State University)

FINDINGS[1]

This paper addresses the problem of the potential effects of projected climatological changes on a river-estuary in the northern Gulf of Mexico. A series of scenarios was developed by other researchers concerning long-term climatological changes due to atmospheric contaminants. Such scenarios involved changes in precipitation/river flow, temperature, and sea level in the region encompassing the Apalachicola-Chattahoochee-Flint drainage system in Alabama, Georgia, and Florida. Using long-term data sets (1972-1987) from the Apalachicola estuary, it was possible to make estimates of potential changes in estuarine productivity and dominant (commercially valuable) populations based on the long-term scenario predictions.

RIVER FLOW

According to models provided by Dr. C. F. Hains (Northport, AL) concerning altered Apalachicola River flow, the following projections (percent losses or gains) could be estimated concerning the response of estuarine functions in the Apalachicola Bay system:

	SCENARIO		
FACTOR	GFDLx2 (%)	GISSx2 (%)	OSUx2 (%)
River flow	-24.2	-6.2	-13.9
Mean salinity	+11.3	+3.1	+6.5
Detritus/nutrients	-10-25	-3-7	-5-15
Detrital food web	-10-25	-3-7	-5-15
White shrimp	-15-35	-5-10	-10-30
Pink shrimp	+10-20	+3-5	+8-20
Blue crabs	-15-35	-5-10	-10-30
Finfishes	-15-35	-5-10	-10-30
Oysters	-20-35	-7-10	-15-30

The above estimates are based on the combined losses of organic carbon and nutrients to the system and on models of increased predation rate on nurserying juveniles of the top dominant estuarine populations due to increased salinity. Oysters would be particularly vulnerable as the salinity increased owing in part to increased predation and disease. These changes are predicated on the assumption that existing populations would not be replaced either by adaptation of existing species or invasion by species new to the region. Such an assumption is somewhat unlikely but remains difficult to estimate in a quantifiable fashion. The increase in pink shrimp would be due to the increased salinity and probable increases in the submerged aquatic vegetation that would accompany the projected changes in river flow and estuarine habitat.

TEMPERATURE

According to the GFDLx2 projections of changes in ambient temperature in the tri-river region, there would be a mean increase of 3-5°C with even greater changes in the mean maximum temperatures of the estuary with time. Such projections are based on the assumption that the estuarine water temperatures would closely follow the air temperatures. Currently, the summer high temperatures of the Apalachicola estuary are close to the upper thermal tolerance limits of most of the dominant (commercially important) species in the system. With such assumptions in mind, the following projections were made based on the established thermal tolerance limits and projected changes of water temperature in the Apalachicola estuary:

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[1] Although the information in this report has been funded wholly or partly by the U.S. Environmental Protection Agency under Contract No. CR-814608-01-0, it does not necessarily reflect the Agency's views, and no official endorsement should be inferred from it.

	SCENARIO		
FACTOR	GFDLx2 (%)	GISSx2 (%)	CURRENT (AMBIENT)* (%)
Blue crab larvae	-100	-100	-77
Blue crab juveniles	-100	-100	-27
Spotted seatrout	-91	-59	-5
Oyster larvae	-91	-59	-5
Adult oysters	n.c.	n.c.	n.c.
Pinfish	-91	-59	-5
Flounder	-91	-59	-5
Redfish	-55	-9	n.c.
Mullet	-5	n.c.	n.c.
White shrimp	-5	n.c.	n.c.

*current loss of habitat use due to summer high temperatures

The above estimates are indications of a loss of estuarine habitat to various commercially important species due to increased water temperature. Currently, juveniles of such species use the estuarine habitat as a highly productive nursery and as a haven from offshore (stenohaline) predators. Such a loss of habitat could not be replaced and the movement of the nurserying populations to offshore regions in the gulf would probably lead to major losses of such populations owing to offshore predation.

The possibility of adaptation and/or replacement by tropical species is quite real but is not easily estimated in the above models. There are currently some very productive tropical estuaries which could serve as models for projected changes in the Apalachicola region. Species such as mullet and white shrimp would not necessarily be lost if temperature is the only variable under consideration. The above projections should also be qualified by the various assumptions implicit in the analysis. However, if the projected temperature changes do occur, there will be some major changes in the structure and productivity of the Apalachicola Bay system with considerable shifts in population dominance and fisheries potential. By analogy, such changes could be projected to other important river-estuaries along the northern gulf coast.

SEA LEVEL RISE

Particulate organic carbon (POC) can be used as an indicator of allochthonous and autochthonous sources of production in the Apalachicola estuary. There are considerable data concerning the production, distribution, and utilization of POC in the Apalachicola drainage system. According to a series of models developed by Dr. Richard Park (Indianapolis, Indiana), various scenarios based on different levels of sea level rise were produced concerning the changes in wetlands currently within the Apalachicola drainage system. Such changes were then translated into overall gains or losses of POC over the period 1987-2100. The following is an estimate of the net input of organic carbon (metric tons/year) in the Apalachicola system based on current conditions and projected sea level rises in the region:

	SCENARIO (by 2100)				
FACTOR	Current	1.2mm/yr	0.5m/yr	1.0m/yr	2.0m/yr
Freshwater wetlands	30,000	26,100	24,000	21,300	4,980
Seagrasses	27,200	28,700	28,800	30,100	31,035
Saltmarsh	46,905	23,500	4,690	940	780
Phytoplankton	233,280	144,640	71,450	58,790	15,160
Total	337,385	222,940	128,940	111,130	51,955

Since there are various other limiting factors that contribute to the recreational and commercial fisheries of the region, it is difficult to estimate the potential losses of such populations. However, the various wetlands serve both as habitats and sources of productivity to the bay system. Any loss of such wetlands would certainly lead to reductions of these estuarine populations. One of the key issues in this estuary concerns the fate of phytoplankton production which is the single most important factor in the high level of productivity in the bay. Such production is largely a measure of freshwater input (e.g., a major nutrient source) to the system and the shallowness of the estuary. The relationship of the phytoplankton and the surrounding wetlands is poorly understood. The above results are predicated in part on the correlation of phytoplankton productivity and the extent of wetland areas; this assumption is further complicated by the lack of an adequate forecast of microbial activity in the estuary (based on increasing losses of wetlands with time). Such twin uncertainties would thus serve to qualify the above projections and would limit any meaningful extrapolation to detailed projections of population response to such changes. Nevertheless, the progressive loss of wetlands to the region would lead to substantive changes in the habitat availability and nutrient/POC budgets in the Apalachicola drainage system.

It is quite likely that a substantial, perhaps even proportional, loss would occur of species that depend on the detrital food webs of the estuary. Such species would include white shrimp, blue crabs, spot, Atlantic croakers, and white seatrout. If, as postulated, the phytoplankton productivity were adversely affected by such losses, species such as anchovies and oysters would also be adversely affected. There would be a replacement of some of the primary productivity of the estuary through increased sea grass bed distribution. This would improve the production of species such as pink shrimp and spotted seatrout.

There are certain shortcomings to the application of correlations, regressions, and multivariate statistical techniques to a complex data set that includes many (diverse) biological variables. Accordingly, this analysis has included various assumptions that must be taken into account when evaluating projected changes in the system according to anthropogenous alterations in the various climatic features of the system. In addition, changes in both water quality and freshwater input to the system due to future human activities associated with municipal developments, industrial expansion, and agricultural activities could significantly alter these projections. An effort has been made to preclude such impacts, although various forms of deterioration due to human activities around the bay have already taken a serious toll on the Apalachicola fisheries. Such effects can be addressed. By contrast, climatological changes would eventually be beyond any efforts of amelioration. It is almost a certainty that the Apalachicola estuary, as it currently exists, would be severely altered by a global warming trend. Although it could be transformed into a productive tropical estuary dominated by seagrass productivity, it is highly likely that there would be an overall decrease in fisheries production and that qualitative changes in the commercially valuable populations would lead to far-reaching changes in fisheries equipment and practices and the availability of seafood to people in the region.

CHAPTER 1

INTRODUCTION

EXISTING DATA BASES: PROJECT SCOPE

This project was designed to evaluate the possible effects of projected long-term changes in the climate (e.g., temperature, precipitation) and associated variables (river flow, sea level changes) related to increased atmospheric CO_2 and other greenhouse gases. The analysis was based on a 15-year data base taken in the Apalachicola Bay system (East Bay, Apalachicola Bay: Figure 1). Data were taken continuously on a weekly to monthly basis from March 1972 through October 1987. Station groups were summed in East Bay (oligohaline) and Apalachicola Bay (mesohaline to polyhaline). The Apalachicola River system (regional) and Tate's Hell Swamp (local) constitute the two major sources of fresh water to the system.

THE APALACHICOLA RIVER-BAY SYSTEM: BACKGROUND

The Apalachicola River is the terminus of a major drainage system that is located mainly in Georgia and Alabama. Seasonal river flow patterns reflect Georgia precipitation and seasonal changes in evapotranspiration rates of the tri-river (Apalachicola-Chattahoochee-Flint) system (Meeter et al., 1987). The winter-spring river peak flows strongly influence various ecologically important estuarine habitat variables (Livingston, 1984a). Emergent and submergent vegetation (freshwater to marine), along with phytoplankton production, provide a year-round source of energy for the system (Livingston, 1983, 1984a) as mediated by river flow and microbial activities (Figure 2). The relationship of river flow and microdetritus is well established as a function of the winter-spring flood events (Table 1). Long-term river flow changes follow interannual periods that approximate 5-7 year cycles (Meeter et al., 1979). Such trends determine salinity changes in the bay, which, in turn, have an important impact on the biological organization of the estuary (Livingston, 1984a).

Organisms at different levels of biological organization follow different seasonal patterns of numerical abundance and species richness (Figure 3). Peak numbers of fishes, dominated by the sciaenids, occur in the winter-early spring months whereas epibenthic invertebrates peak in the late summer and fall months and the infaunal macroinvertebrates peak in the winter. Specific dominant fish and invertebrate populations, some with great commercial value, follow highly individualistic patterns that vary seasonally and yearly (Figures 3, 4, and 5). Most such populations spawn offshore (Figure 6). Others, such as oysters, spend their entire lives in the estuary. As a consequence of the river input and basic climatological conditions, the Apalachicola system produces a major fishery with a dock value of oysters, penaeid shrimp, and finfishes of over $12 million in 1980. According to Prochaska and Mulkey (1983), application of multipliers to such output would increase the regional value of such fisheries to between $17 and $23 million for that year. Projected values of the estuary by the year 2000 have varied from $34 million (Colberg et al., 1968) to $67 million (Mulkey and Maturo, 1983). The Apalachicola Bay system also serves as a habitat for various rare, endangered or threatened species as well as a major flyway for diverse birds (Livingston and Joyce, 1977).

The basic model (Figure 7) for the ongoing analysis of the Apalachicola system indicates the extreme complexity involved in any prediction of future changes based on climatological alterations. Because of this complexity, there are various assumptions that must be made when applying statistical models to the field data. Such assumptions severely restrict a truly quantitative approach to the problem so that the following analysis is basicly a first-cut attempt to project relatively broad responses to the projected changes in the system based on the climatological scenarios provided by the Environmental Protection Agency.

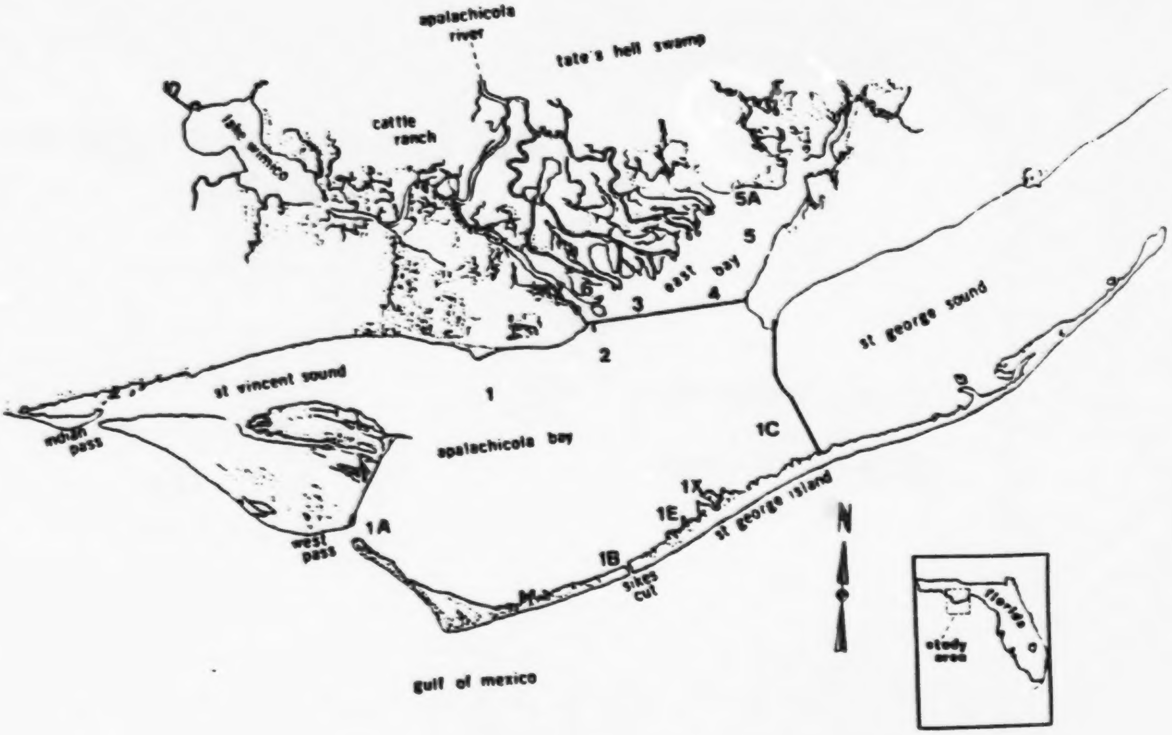
apalachicola river
tate's hell swamp
lake wimico
cattle ranch
5A
5
east bay
6
3
4
2
1
st vincent sound
st george sound
indian pass
apalachicola bay
1C
1X
1E
1A
west pass
1B
sikes cut
st george island
N
gulf of mexico
florida
study area

Figure 1. The Apalachicola estuary showing local drainage areas and the river model showing relationships of detrital flow through the Apalachicola estuary (according to Livingston, 1984a). Also shown are sampling stations used in the long-term monitoring program.

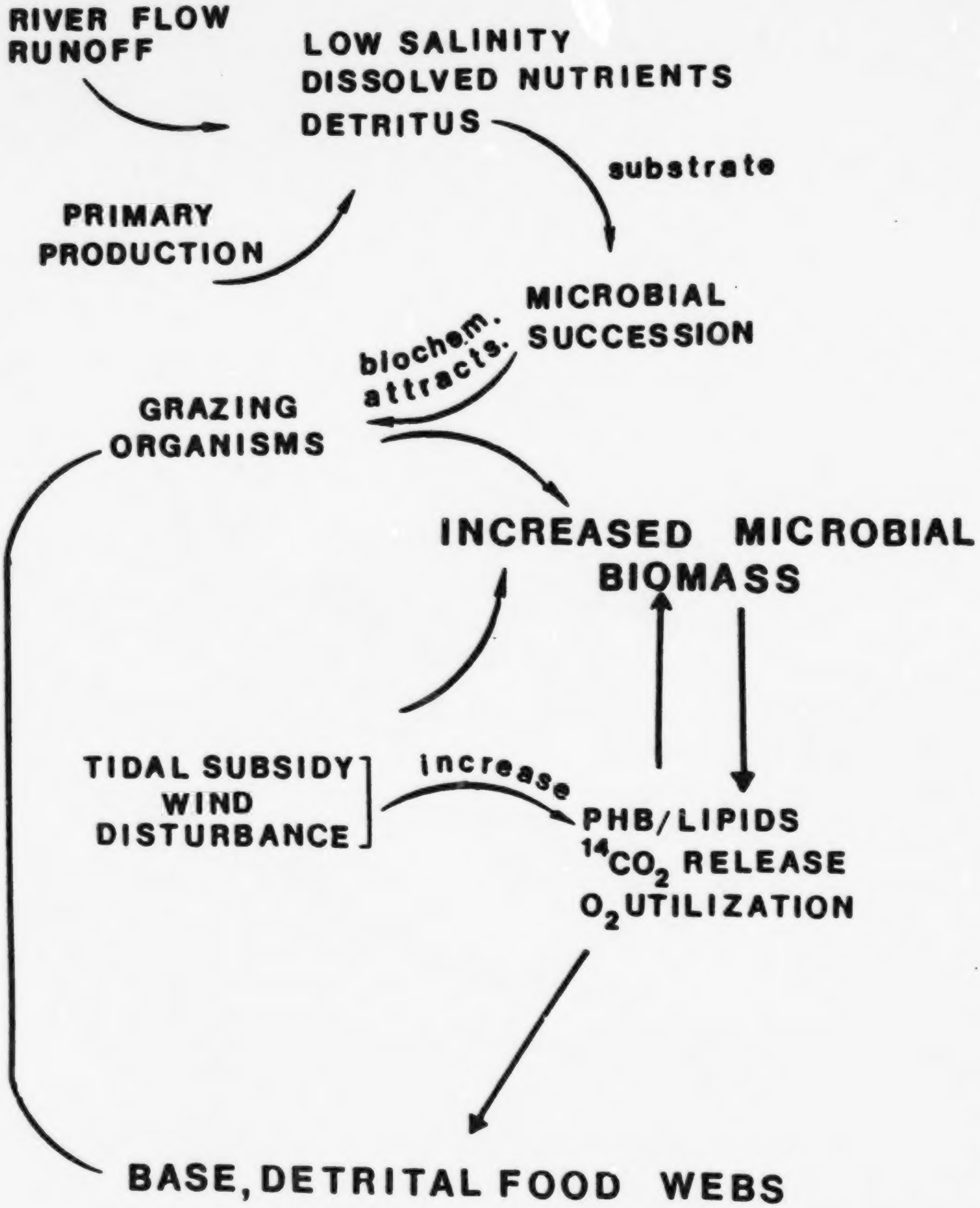
RIVER FLOW
RUNOFF
LOW SALINITY
DISSOLVED NUTRIENTS
DETRITUS
substrate
PRIMARY
PRODUCTION
MICROBIAL
SUCCESSION
biochem.
attracts.
GRAZING
ORGANISMS
INCREASED MICROBIAL
BIOMASS
TIDAL SUBSIDY
WIND
DISTURBANCE
increase
PHB/LIPIDS
$^{14}CO_2$ RELEASE
O_2 UTILIZATION
BASE, DETRITAL FOOD WEBS

Figure 2. Model showing relationships of detrital flow through the Apalachicola estuary (according to Livingston, 1984a).

Table 1. Linear Regression (log/log) of Total Microdetritus (Ash-Free Dry Weight) and Riverflow (m^3/sec) by Month/Year by Season (8/75-4/80)

Station 7 (Surface)	R	R^2	α (significance)
December-February	0.70	0.49	0.00188
March-May	0.77	0.60	0.00057
June-August	0.08	0.01	0.39863
September-November	0.48	0.23	0.03469
Station 7 (Bottom)			
December-February	0.77	0.60	0.00037
March-May	0.55	0.30	0.02253
June-August	0.08	0.01	0.40243
September-November	0.21	0.04	0.22867
Station 8 (Mid-depth)			
December-February	0.64	0.40	0.00570
March-May	0.68	0.46	0.00397
June-August	0.35	0.12	0.11809
September-November	0.19	0.04	0.25542

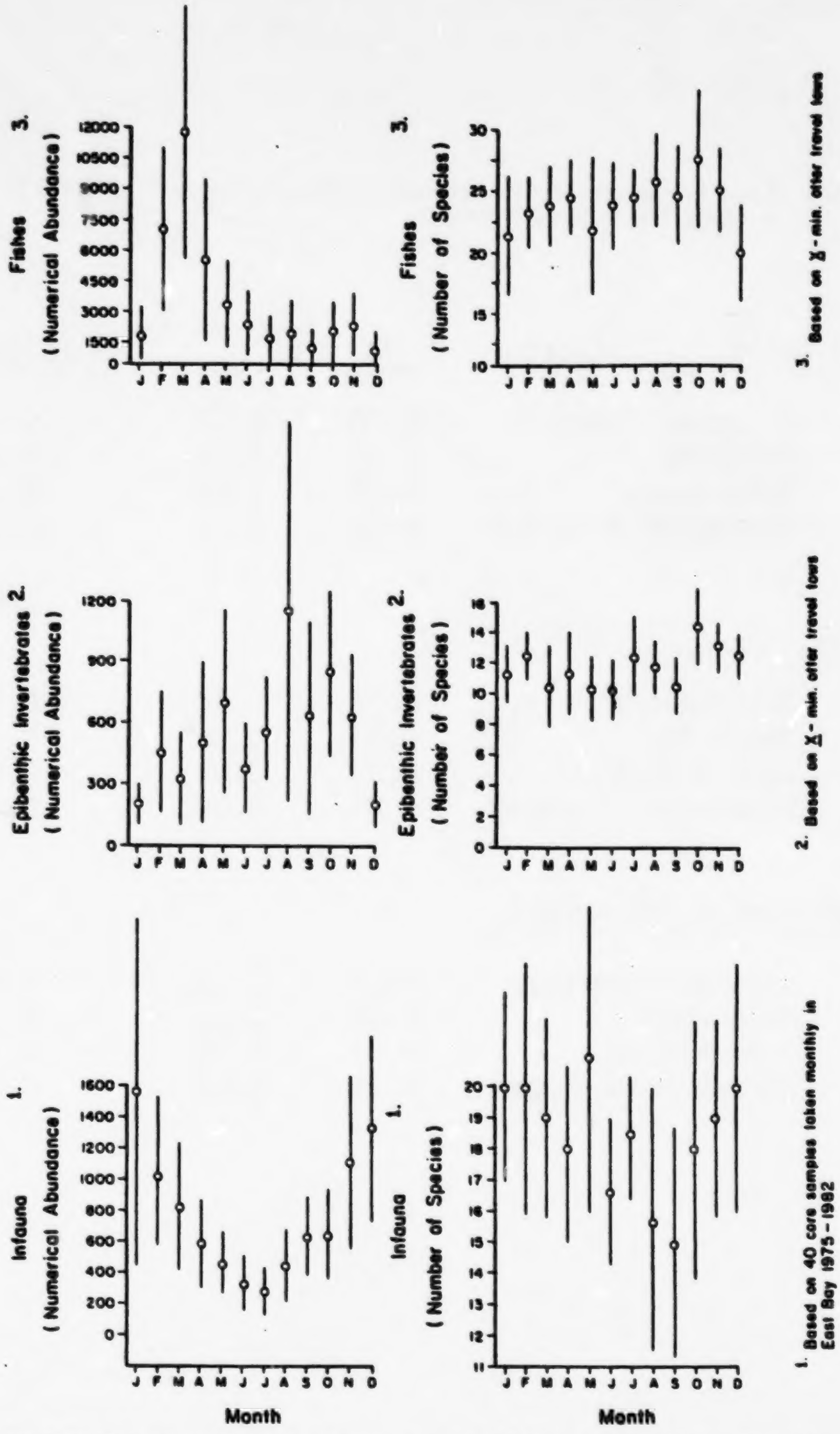

Figure 3. Mean monthly values of numerical abundance and species richness of infaunal macroinvertebrates, epifaunal macroinvertebrates, and fishes taken in the Apalachicola estuary from 1972-1982.

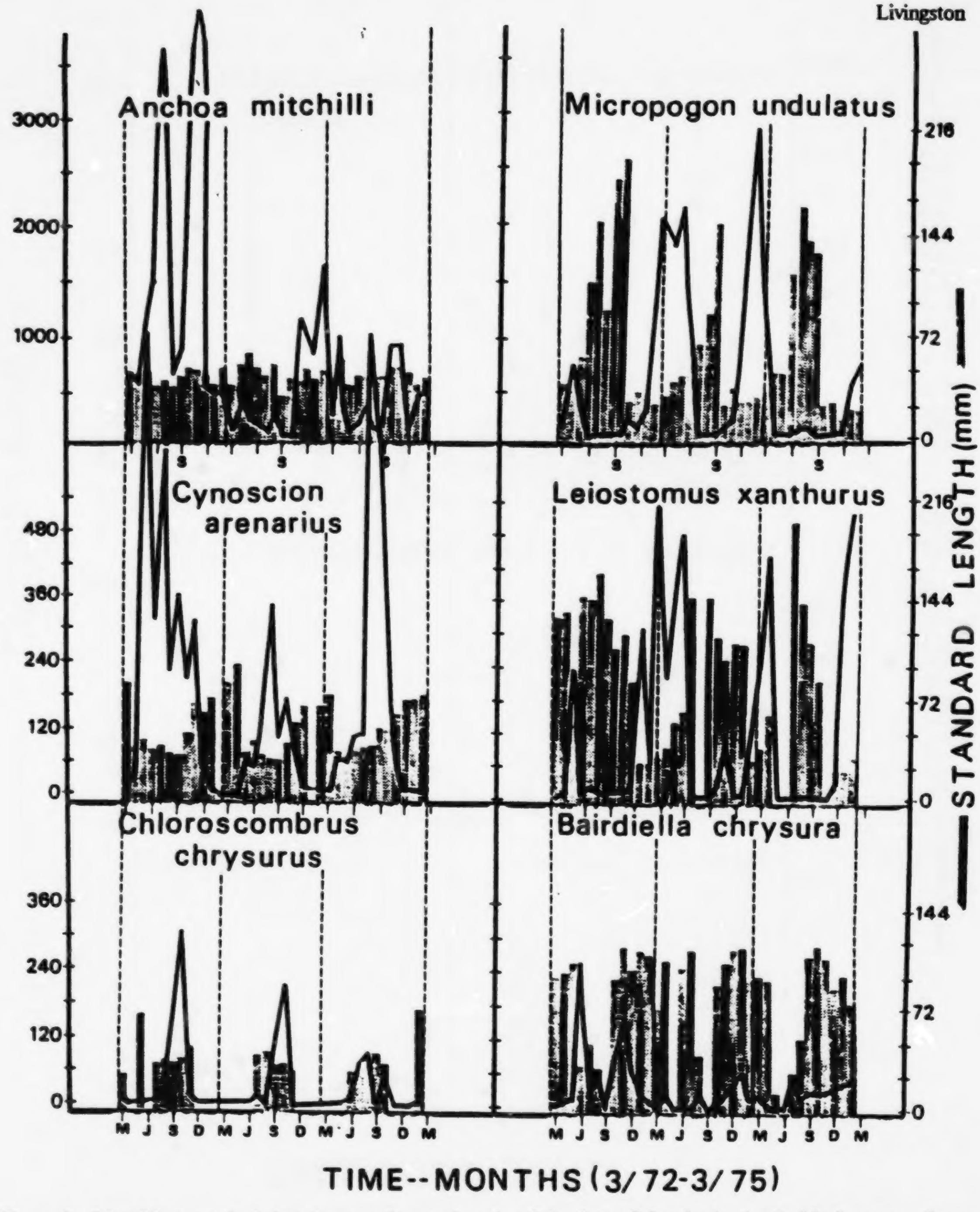

Figure 4. Monthly numerical abundance and mean lengths of dominant fishes in the Apalachicola estuary from 3/72-3/75. Populations include anchovies, croaker, spotted seatrout, spot, Atlantic bumper, and silver perch.

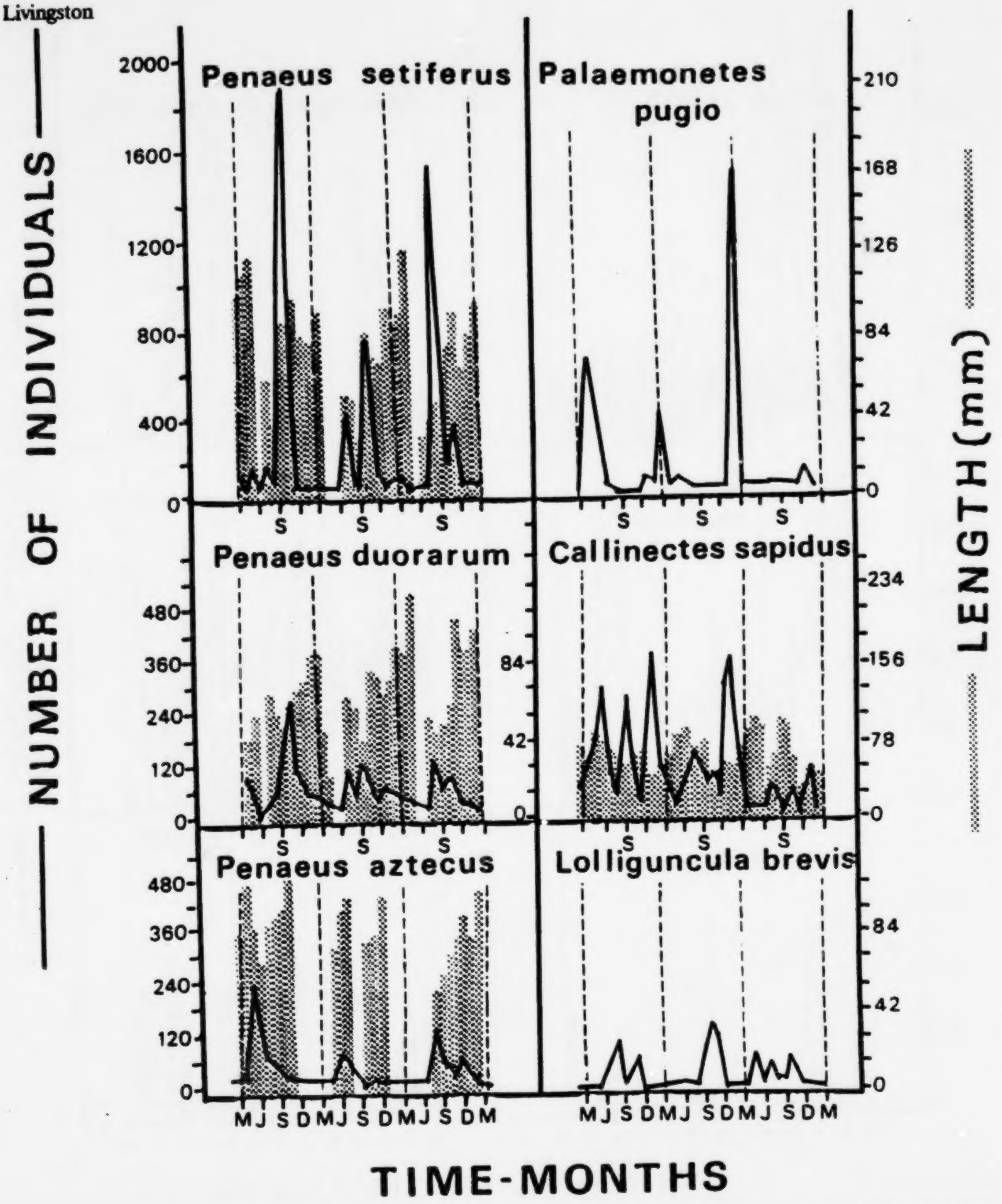

Figure 5. Monthly numerical abundance and mean lengths of dominant invertebrates in the Apalachicola estuary from 3/72-3/75. Populations include white shrimp, paleomonetid shrimp, pink shrimp, blue crab, brown shrimp, and least squid.

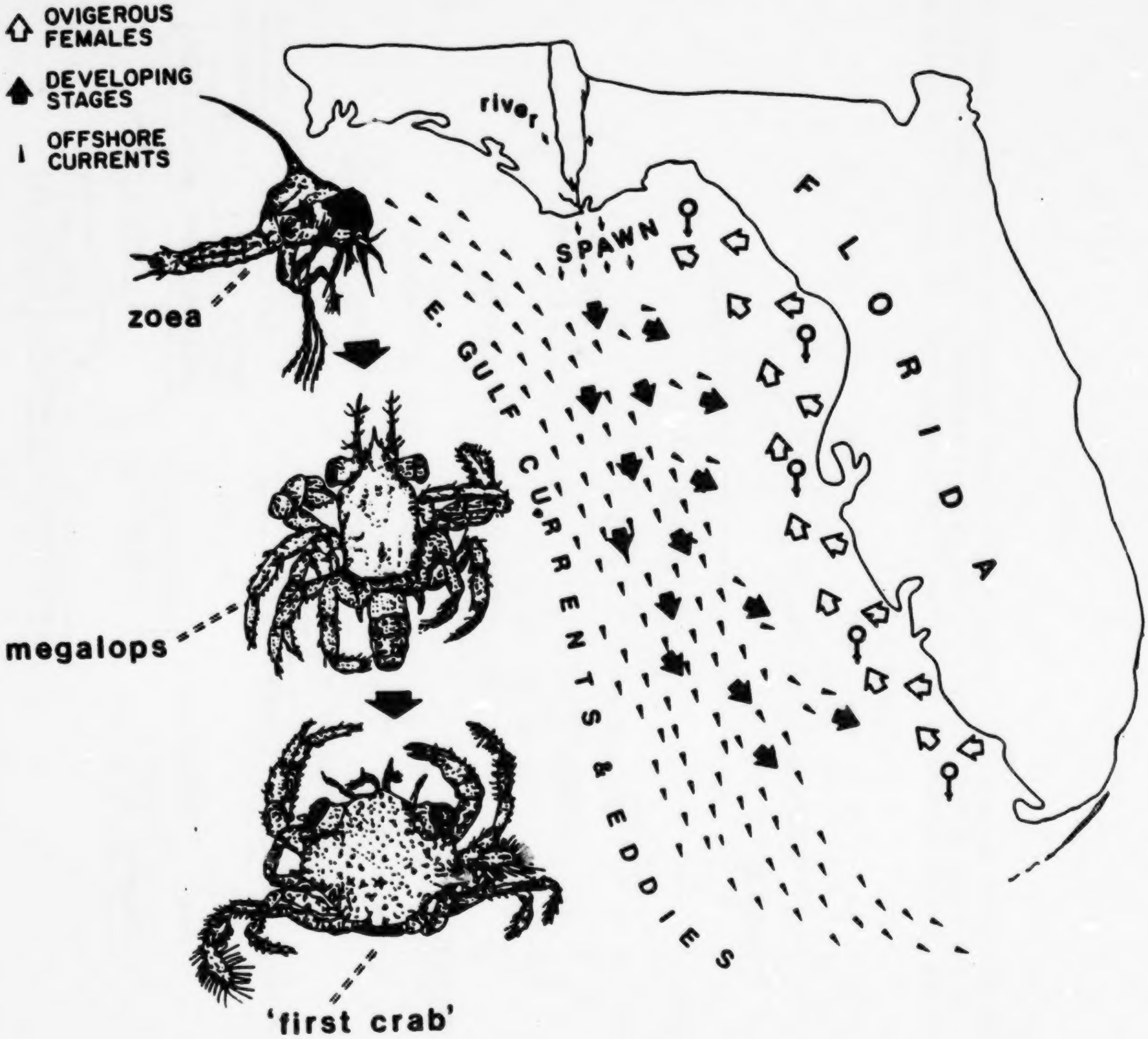
OVIGEROUS FEMALES
DEVELOPING STAGES
OFFSHORE CURRENTS
river
SPAWN
FLORIDA
E. GULF CURRENTS & EDDIES
zoea
megalops
'first crab'

Figure 6. Schematic diagram of the life history movements of blue crabs along the Florida gulf coast. Ovigerous females move north to spawn in the waters offshore from the Apalachicola-Ochlockonee River systems. After spawning, the young move south, possibly using existing currents in offshore areas. The young crabs nursery in inshore waters such as the Apalachicola estuary.

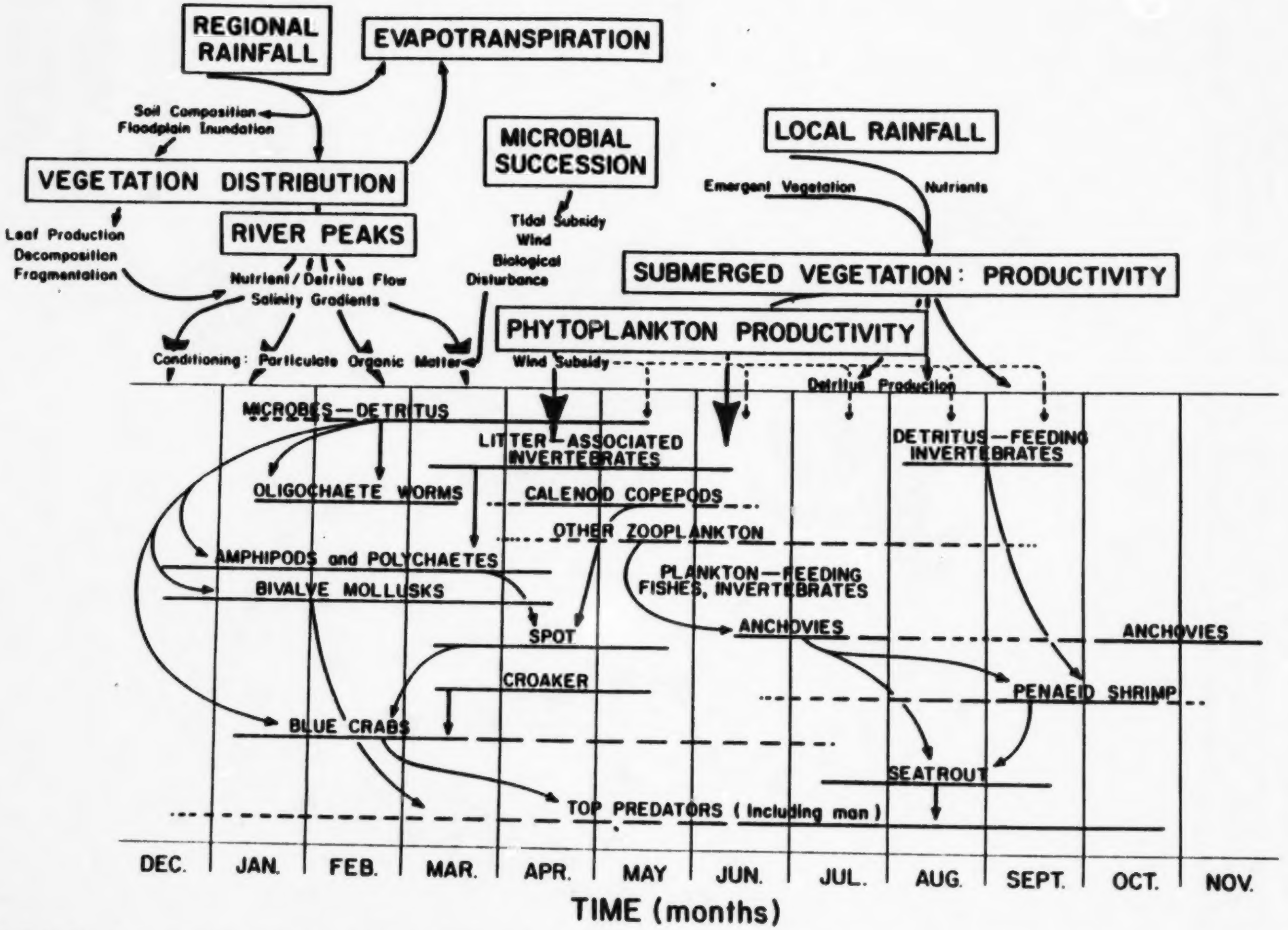
REGIONAL RAINFALL
EVAPOTRANSPIRATION
Soil Composition
Floodplain Inundation
VEGETATION DISTRIBUTION
Leaf Production
Decomposition
Fragmentation
RIVER PEAKS
Nutrient / Detritus Flow
Salinity Gradients
MICROBIAL SUCCESSION
Tidal Subsidy
Wind
Biological
Disturbance
LOCAL RAINFALL
Emergent Vegetation
Nutrients
SUBMERGED VEGETATION : PRODUCTIVITY
PHYTOPLANKTON PRODUCTIVITY
Conditioning : Particulate Organic Matter
Wind Subsidy
Detritus Production
MICROBES—DETRITUS
LITTER—ASSOCIATED INVERTEBRATES
DETRITUS—FEEDING INVERTEBRATES
OLIGOCHAETE WORMS
CALENOID COPEPODS
OTHER ZOOPLANKTON
AMPHIPODS and POLYCHAETES
PLANKTON—FEEDING FISHES, INVERTEBRATES
BIVALVE MOLLUSKS
SPOT
ANCHOVIES
ANCHOVIES
CROAKER
PENAEID SHRIMP
BLUE CRABS
SEATROUT
TOP PREDATORS (Including man)
DEC.
JAN.
FEB.
MAR.
APR.
MAY
JUN.
JUL.
AUG.
SEPT.
OCT.
NOV.
TIME (months)

Figure 7. Model showing possible relationships of different variables in the Apalachicola estuary. The model incorporates the various sources of energy to the system in addition to the various forms of climatological factors that control the biological processes in the estuary. The dominant populations are shown during months when organisms reach peak numbers in the estuary.

CHAPTER 2

METHODS

GENERAL APPROACHES

Ideally, to develop the appropriate models for an assessment of the potential impacts of projected climatological changes on the Apalachicola estuary, we would need to build a comprehensive model of the entire ecosystem and food web. Such a model would have to involve both time series analysis and simulations that would incorporate observed changes, temporal dynamics, and species interactions. Such an effort is outside the scope of this effort. Instead, we have used simple regression and multivariate analyses to evaluate the responses of various portions of the estuary over 15 years of continuous observation. Thus, this analysis can be considered preliminary to a more comprehensive modeling effort using more sophisticated modeling techniques.

The Apalachicola river-estuary has been studied for the past 18 years (15 years of field work, 3 years of analysis) by scientists associated with the Center for Aquatic Research and Resource Management (Florida State University). A complete list of the data available for this analysis along with a summary of quality assurance and analytical protocols is given in Appendix I (appendices are available from the U.S. Environmental Protection Agency). Background publications are presented in Appendix II. The field data base consists of monthly collections of important variables taken over a 13-year period (Table 2). The oyster populations were studied for an additional 2 years. The exact methods of data collection have been given in Livingston (1983, 1984a) and associated papers (Appendix II). Apalachicola River flow was a primary variable in the evaluation. During the study, there was a major river flooding episode (1973-1975) and a prolonged drought (1980-1981). The response of the estuary to these extreme conditions was evaluated relative to background levels of river flow.

The Apalachicola estuary was the subject of the present analysis which included station-specific data (Figure 1), grouped stations (East Bay, Apalachicola Bay), and estuary-wide averages. A series of scattergrams was developed using the monthly field data. Various transformations were made to determine the best fit for a normalized distribution when running the different statistical analyses. Such transformed data were used for all calculations. Moving averages were constructed to eliminate specific aspects of the short-term variability. Multiple correlation matrices were constructed using all variables. Such correlations were made with and without lags of key factors. Regressions were used to determine specific relationships among the climatological, physical, chemical, and biological variables. In this way, certain relationships between and among the different dependent and independent variables were ascertained.

SCENARIOS FOR EFFECTS MODELING

This analysis required scenarios of future temperature, river flow, and wetland loss. Estimates of wetland loss due to sea level rise were provided by Park (Volume B). Hains (Volume A) provided scenarios of future river flow based on climate model projections of temperature, rainfall, and evapotranspiration. In addition, we developed scenarios of river flow based solely on precipitation scenarios from the models. The climate change scenarios provided by the Environmental Protection Agency (EPA) were used as inputs to the effects models. We used the GISS $2CO_2$ and the GFDL $2xCO_2$ models for our calculations. Data provided by the EPA were used (e.g., means by month) for surface air temperature (°C) and precipitation (mm day^{-1}) levels in our region (GFDL, 82.5°W, 33.33°N; GISS, 80°W, 35.22°N) (see Table 3). Rainfall data (ambient) were used from Columbus, Georgia (85°W, 32.5°N). The NOAA data from Columbus, Georgia included minimum temperature (°F), maximum temperature (°F), and precipitation (0.01 inch) on a daily basis from 1/1/48 to 12/31/83. Data bases were constructed for the entire data set and for data from 1951 to 1980. The period 1/84 to 7/84 was added from our own data base so that the final numbers could then be applied Table 2.

Table 2. Data Base Used in the EPA Climate Project

I. Climatological features
  a. temperature
  b. precipitation
  c. river flow

II. Physical/chemical factors
  a. temperature
  b. salinity
  c. dissolved oxygen
  d. pH
  e. color
  f. turbidity
  g. oxygen anomaly

III. Productivity features
  a. microparticulates
  b. macroparticulates
    1. leaf matter-detritus
    2. sea grasses

IV. Biological variables
  a. infaunal macroinvertebrates
    1. numbers
    2. biomass
  b. epibenthic macroinvertebrates
    1. numbers
    2. biomass
  c. fishes
    1. numbers
    2. biomass
  d. fisheries

Table 3. Decadal Monthly Values for GISS and GFDL Models at Georgia Grid Points*

MONTH- TYPE	1	2	3	4	5	6	7	8	9	10	11	12
GISST	1.0128	1.0122	1.0218	1.0114	1.0149	1.0128	1.0102	1.0073	1.0173	1.0183	1.0128	1.0097
GISSP	1.0873	1.2439	1.2369	1.1783	1.1392	1.3120	1.4133	1.2309	.7970	.9699	.7464	.9169
GFDLT	1.0122	1.0127	1.0151	1.0184	1.0127	1.0209	1.0245	1.0148	1.0206	1.0161	1.0190	1.0190
GFDLP	.7689	1.0537	1.1465	1.1493	.8283	.4408	.6514	.4257	.8082	.6737	1.0781	1.0716

T = temperature
P = precipitation

* Values for Columbus, GA (85•N, 32.5•W)

to our long-term (Apalachicola) data base, which ran from March 1972 through July 1984. The calculated adjustment factors for the GISS and the GFDL models were then applied to the data for the predicted effects of the doubled CO_2 scenarios. There was a different adjustment factor for each month in the respective models for the surface air and precipitation values:

Temperature: [(average TX 273.15)Xadj] - 273.15
Precipitation: total precipitationXadj.

The ambient data (precipitation, temperature maxima and minima) were then compared with the projected (GISS,GFDL) results for the different time periods (1/1/48-12/31/83, 1/1/51-12/31/80, 3/1/72-7/31/84).

The GISS precipitation model predicted generally higher levels of precipitation, whereas the GFDL precipitation model predicted somewhat less. The study period (3/72-7/84) appears as part of an overall (long-term) decline in precipitation with the drought of 80-81 shown clearly. When viewed as differences in the respective moving averages, the GISS model predicts average increases of a little less than 2 cm/month whereas the GFDL model predicts average decreases of just under 2 cm/month.

Using the previously developed regressions linking precipitation to Apalachicola River flow at Blountstown, Florida, we used the various data projections to calculate new river flows (based on the GISS and GFDL scenarios). The regressions were also used to calculate the salinity regimes in the Apalachicola estuary over the period of observation (3/72-7/84). The various relationships (e.g., correlation, regression) were then calculated for the different physical, chemical, and biological variables (Table 2). Regressions were applied to the model river data to get projected river flows according to the EPA models. Such models were then applied to the various biological variables taken in the Apalachicola estuary (3/72-7/84).

The various biological indices (e.g., population, community) were then analyzed in the existing data base with particular attention to responses to changes in river flow, salinity, and temperature. Such data were supplemented with reviews of the literature concerning temperature ranges and optima for estuarine dominants and commercially valuable species. Using the above-outlined scenarios of climate change and the combination of quantitative and qualitative analyses of biological response to such variables, a series of tentative conclusions was outlined relative to the ranges of predicted events.

CHAPTER 3

RESULTS

RIVER FLOW SCENARIOS

An application of the models to the Columbus, Georgia precipitation data over the 13-year study period indicated differential seasonal differences. The GFDL model predicted considerable reductions in rainfall that are most evident during January and summer-fall periods. The GISS model predicted general increases in precipitation that approach 8 cm in extreme cases. The overall trends showed differences approaching or ± 2 cm in the two models. There is a pronounced reduction in the differences in the GFDL model during the latter half of the record, which reflects general reductions of Georgia precipitation.

The Apalachicola River flow data (Figure 8) show the extreme high and low level years quite clearly. Generalized peak flows were most consistent during 1973; the overall drought months (20) occurred during the period 1980-1981. These overall trends are evident in the 12-month summed data. Peak flows occurred during 1973; this was followed by an overall decline after a secondary peak in 1975. This decline ended in the drought year 1981, which was followed by a moderate increase in 1983-1984. During the 24-month period of 1980-1981, 15 months had monthly river flow averages that were below 500 m^3; this was the most prolonged drought during the period of observations (3/72-7/84). The seasonal distribution of such flow conditions showed that, with the exception of the spring of 1979 and 1980, the overall decline of Apalachicola River flow from 1973 to 1981 was real. Summer-fall periods varied the least with the general trend following the winter-spring flows.

A statistical examination of the relationship between Apalachicola River flow and rainfall (Columbus, Georgia; Apalachicola, Florida; East Bay, Florida) indicates that only Georgia rainfall is significantly associated with such river flow ($r^2 = 0.207$, $p < 0.0001$). There is a close relationship of Georgia rainfall and river flow with some evidence of an averaged lag period of 3 months between the Columbus, Georgia, precipitation and the Blountstown, Florida, Apalachicola River flow. Using the established regression between these two variables, the GISS and GFDL models were run to determine the new river flow conditions under the projected precipitation change. The GISS model predicted an average increase in river flow conditions from the observed (from 722.277 to 827.307 m^3 per second). The minimal levels were higher, whereas the maximum levels were lower. The GFDL model, however, predicted an average decrease (from 777.369 to 722.277 m^3 per second). Minimal values increased, while maximal values were less. In both cases, the extremes were moderated with less high flow in the winter-spring months and higher flow rates during the summer-fall months.

The mean salinities for the Apalachicola estuary (Figure 10) show trends that are related to the above described long-term changes in meteorological conditions. During the 1980-1981 period, 10 mean salinity points were above 15 ppt, the highest such number in the entire survey. The spring flooding in 1979 and 1980 was particularly effective in driving down the bay salinity levels. A 12-month moving average of the data (Figure 11) shows the general decline in salinity prior to the major increases during the drought of 1980-1981. The spatial distribution of such salinity during the drought showed considerable vertical stratification and distinct differences between the oligohaline East Bay and the meso- to polyhaline Apalachicola Bay (Livingston, 1984a). Such differences are crucial to an understanding of the biological organization of this estuary.

The GISS and GFDL models were run with the Apalachicola River flow regressions (Table 4). According to the GISS model, mean salinity (overall) is decreased from 8.9 ppt to 8.4. Model projections of minimum and maximum salinities are considerably higher and lower, respectively, when compared to the actual data. According to the GFDL model, mean salinity increases from 8.9 ppt to 9.2 ppt. The relationship of the minimum and

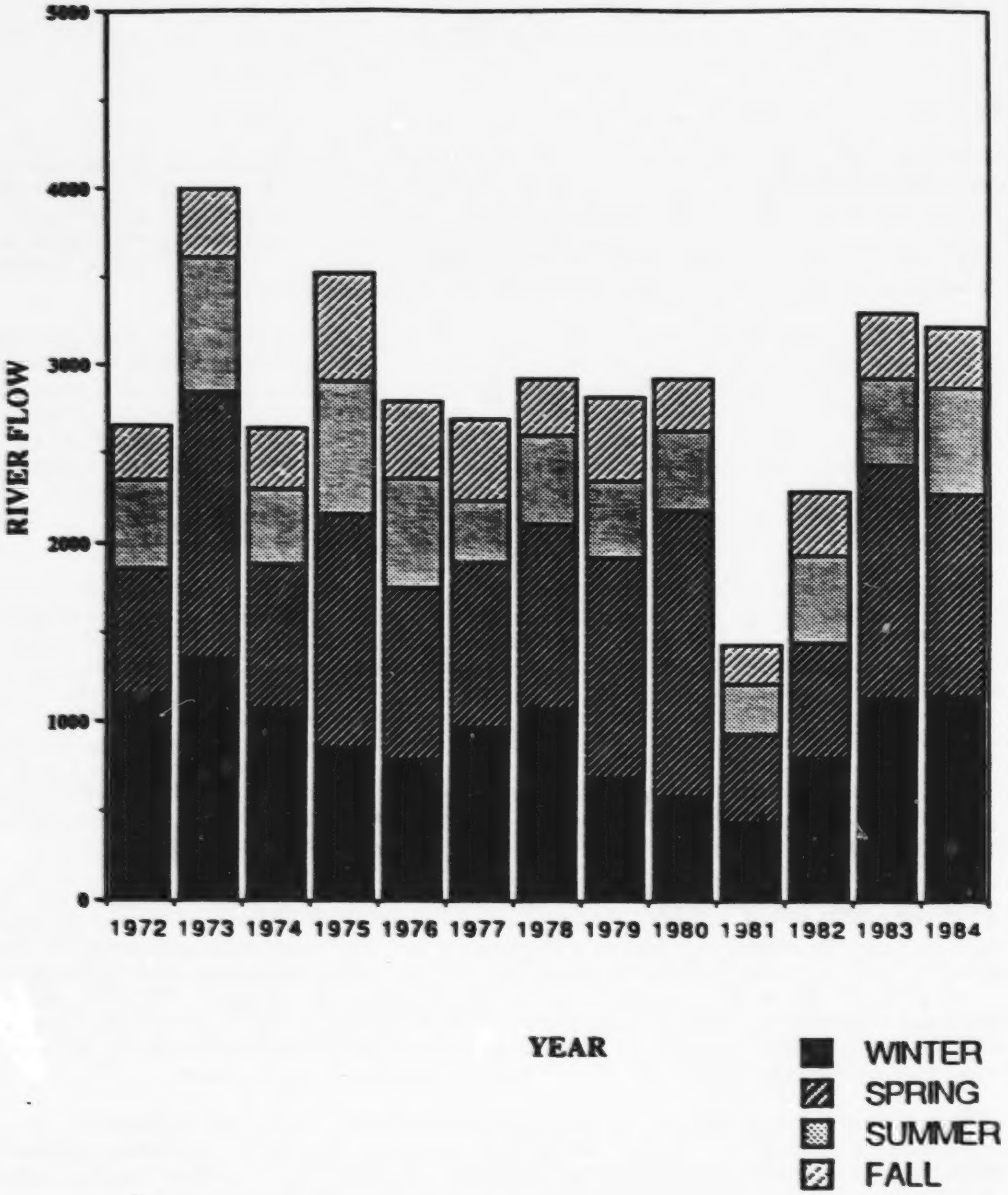

Figure 8. Annual quarterly (seasonal) means of Apalachicola River flow from 1972-1984. Data show peak levels during 1973-1975 and the drought during 1981-1982.

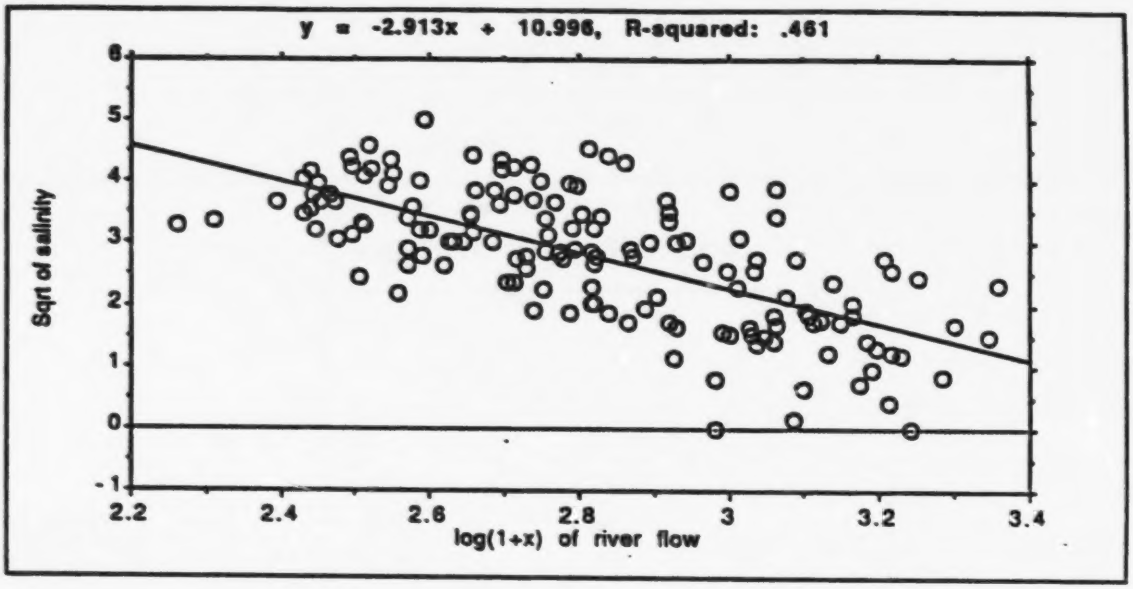

Figure 9. Regression of the square root of salinity on the log (x+1) Apalachicola River flow (monthly data; 1972-1984). Salinity represents monthly means of all stations in the estuary, surface, and bottom. River flow data include monthly means taken at Blountstown, Florida. There is a strong inverse relationship between these two variables.

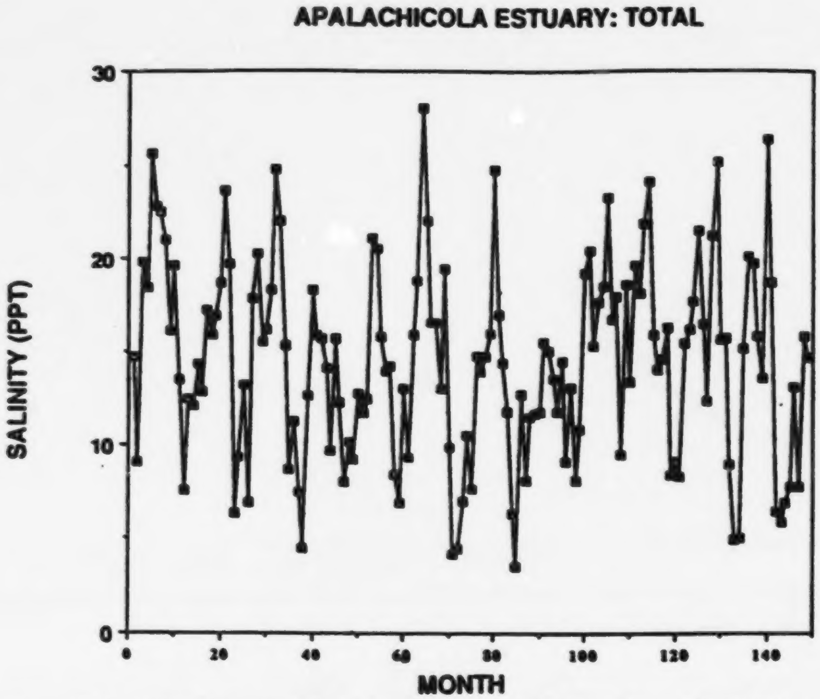

Figure 10. Average salinity data taken monthly in the Apalachicola Estuary from 3/72-7/84. Monthly means of all stations (surface and bottom) were used for this analysis.

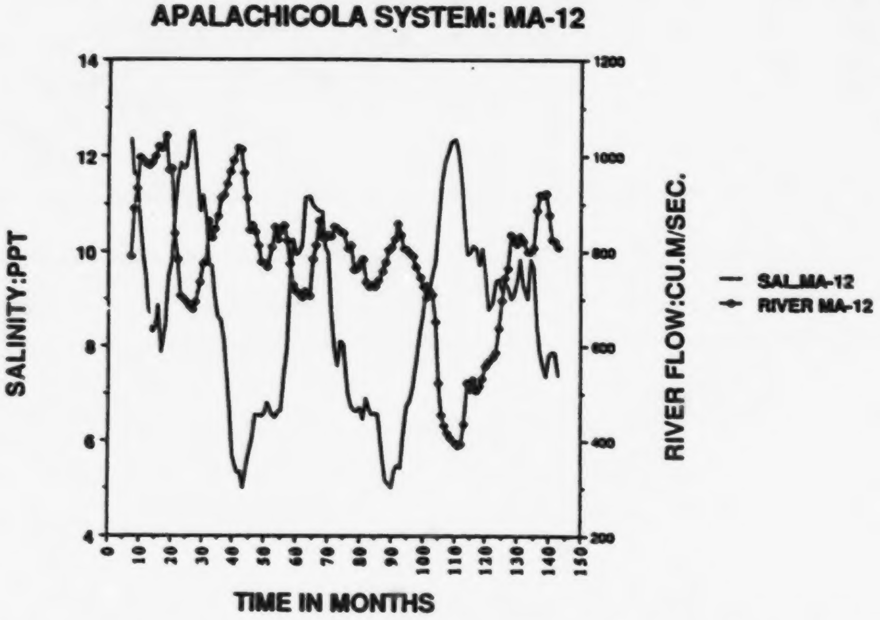

Figure 11. Twelve month moving average of river flow and average salinity data taken monthly in the Apalachicola estuary from 3/72-7/84. River slow data are monthly mean values taken at Blountstown, Florida. Salinity data include monthly means of data taken from all estuarine stations (surface and bottom).

Table 4. A: **Regression Statistics Concerning the Relationship of Apalachicola River Flow and Mean Salinity Values in the Apalachicola Estuary**

	Mean	Std. Dev.	Min.	Max.	Range	# cases
River flow	8.9	5.7	0	25.0	25.0	149
GISS	8.4	2.1	2.2	12.0	9.8	149
GFDL	9.2	1.9	2.7	12.0	9.2	149

B: **Regression Statistics Concerning the Relationship of Apalachicola River Flow and Salinity According to Hains' GISS, GFDL, and OSU Projections**

Data projections are made monthly from January, 1948 through July,1984.

	Mean	Std. Dev	Min.	Max.	Range	# cases
Nat. flow	9.9	3.1	0	13.7	13.7	372
GISS	10.2	3.0	0	13.9	13.9	372
GFDL	11.0	3.1	0	14.3	14.3	372
OSU	10.6	2.5	2.1	13.6	11.5	372

maximum salinities parallels that described for the GISS model. In both models, the peaks and valleys of the salinity levels are largely eliminated. Such results are obvious artifacts of the methods (e.g., regression) used to arrive at the final projections.

River flow projections according to the Hains models indicated a 24.2% decrease in river flow projected by the GFDLx2 model; lessor decreases were noted with the GISSx2 and OSUx2 models. Such changes in river flow would result in a maximum (GFDLx2) increase of 11.3% of the mean salinity in the Apalachicola estuary (Table 4). Such a change would result in generally higher salinities over time with such increases particularly evident in the summer-fall highs. These differences approach a range of +/-10 ppt in a given month. The differences can go either way over the time of projection although most such differences are positive.

Water Quality Changes

Various water quality factors in the Apalachicola estuary are strongly affected by river flow conditions (Livingston, 1984a). Mean turbidity values for the estuary reflect the major river flooding during the winter-spring of 1973. The decreased turbidity in the following years indicates reduced flows; the same is true for the latter portions of the 1980-1981 drought. Color levels follow a different pattern with the lowest levels noted during the 1980-1981 drought. The long-term pattern of Secchi data reflects the above-noted episodes of river flow; the highest Secchi readings (e.g., the clearest water) were noted during the drought period. The oxygen anomaly (AO_2), denoted as excesses of dissolved oxygen above or below solubility, has been associated by some authors with productivity.

$AO_2 = O_2 - O_2$ (calculated from the equation of Weiss 1970 as the difference between the measured oxygen and the oxygen solubility at ambient temperature and salinity)

Analyses of the oxygen anomaly in the Apalachicola estuary (Justic and Livingston, in review) indicate a weak positive association with temperature and a weak negative association with river flow in East Bay and Apalachicola Bay. Relatively high correlations were noted in surface waters in both systems between the oxygen anomaly and salinity; lesser such correlations were noted at the bottom. This would indicate possible increases in phytoplankton activity during periods of high temperature, low river flow, and high salinity. Such trends would be consistent with known patterns of phytoplankton productivity in the estuary (Livingston, 1983, 1984a). This would indicate that reduced river flow could be associated, at least during the short term, with increases in phytoplankton productivity.

River-driven estuarine systems such as the Apalachicola are often dependent, to a considerable degree, on organic detritus that comes from upland wetlands and submerged aquatic vegetation. From January 1976 through March 1982, a survey was conducted to determine the relative amounts of POM that were occurring in the bay. Water from the lower end of the river was pumped through a series of sieves and analyzed for organic carbon (total particulate organic carbon; sized fractions of POM from 2 mm to 0.1 µm). Such matter forms the substrate for microbial activity that is directly linked to the detritus-based food webs of the bay (Livingston, 1984a, unpublished data with D.C. White). The regional rainfall (Georgia and Florida) along with the Apalachicola River flow was low from June 1980 through December 1981. Although concentrations of TPOC were high during certain months of the drought, the overall mass transport of TPOC was relatively low throughout this period. Such mass flows are directly related to river flow levels. Most of the POC coming down the river is composed of very fine particulate organic carbon (VFPOC). The TPOC (concentration and mass flow) was negatively associated with temperature and positively associated with river flow. In East Bay during the drought, salinity was uniformly high. Color was low and Secchi readings were high, and the peak oxygen anomaly readings occurred at this time. The bay salinity was very high during the drought; in most months, mean levels exceeded 20 ppt. Once again, Secchi readings were high and color was low. The POM was particularly low in all its forms during the drought. A summary of these data (Appendix VI) indicates that all forms of POM except the SAV were lower during the drought than at other times when rainfall/river flow rates were higher.

The data indicate that, during periods of low rainfall and river flow, the bay becomes clearer, more saline, and has less POM derived from the winter/spring winter flooding. Such conditions could lead to short-term increases in phytoplankton productivity and the proliferation of submerged aquatic vegetation. When viewed on the basis of overall productivity in the bay, there would be a net increase in the spring to fall productivity at first. However, as nutrients became limiting, there would probably be a trend toward lower phytoplankton production and higher SAV production with conditions resembling the seagrass systems of Apalachee Bay (see Livingston, 1986, for a review of the long-term conditions in this system). Those forms that utilize the estuary in the winter-early spring months (sciaenid fishes) or are dependent on phytoplankton productivity (oysters) would be adversely affected by such conditions. Other species, such as the spotted seatrout (Cynoscion nebulosus) and other forms that depend on the seagrass beds would be favored. The data indicate that if river flows are reduced, the major commercial fisheries of the Apalachicola Bay system would be adversely affected whereas the sports fisheries associated with increased seagrass beds would be enhanced. Prolonged increases in regional rainfall and river flow would have the opposite effect.

Long-Term Biological Changes

Salinity is one of the most important ecological variables in estuarine systems. Low and variable salinities tend to reduce the number of offshore species that can live in the estuary; this makes this kind of system a haven or sanctuary for those species that are euryhaline. The complex community structure and food web patterns that are associated with the high productivity and salinity relationships tend to explain why estuaries are important centers of sports and commercial species. However, such complexity precludes the direct use of correlation/regression analyses to predict specific relationships with estuarine habitat features and important forcing functions. Thus, the following is a preliminary approximation of such relationships since it is based primarily on such statistics.

The overall trends of the infaunal macroinvertebrates are shown in Figure 12. Such data indicate that both species richness and numerical abundance vary together ($r^2=0.603$; $p < 0.0001$). When viewed as a function of river flow (Figure 12), the increases in both indices are associated with the drought period even though the general trends are not significantly correlated. Salinity does not appear to be a major factor in these trends, at least directly; increased salinity could have an indirect effect by allowing predators into the area. In a series of experiments (Mahoney and Livingston, 1982; Livingston, in press; Livingston, unpublished data), it was shown that direct predation has little impact on the infaunal associations in East Bay. There is some indication, according to these analyses, that the East Bay infaunal assemblages are physically controlled by temperature and river/rainfall conditions. The considerable drop in numbers after the drought could be an indication of a delayed response to the reduction of detrital matter associated with the drought.

The epibenthic invertebrates represent some of the most important species in the estuary in terms of their commercial value. The two top dominants are the white shrimp (Penaeus setiferus) and the blue crab (Callinectes sapidus) which together with the oyster (Crassostrea virginica), provide the basis for a multimillion dollar industry in the region. Invertebrate numbers were highest during the years of highest rainfall, whereas invertebrate biomass was highest during the 1980-1981 drought (Figure 13). The penaeid shrimp tended to have peak numbers during periods of high river flow (Figure 14), whereas the blue crabs were in the highest numbers during the drought (Figure 14). The regressions of invertebrate indices with physical/chemical factors (Appendix VI) indicate reductions of numbers and biomass with increasing river flow, POC influx, and decreasing salinity.

The fish associations of the Apalachicola estuary are well developed with numerical dominants including the bay anchovy (Anchoa mitchilli), spot (Leiostomus xanthurus), croaker (Micropogonias undulatus), and white seatrout (Cynoscion arenarius). The numbers of such species usually peak during winter-early spring periods. The 13-year trend (Figure 15) indicated the highest numbers of fishes during the 1980-1981 drought, whereas fish biomass was highest during and immediately following the period of high river flooding. Fish biomass tended to follow the long-term trends of river flooding with particularly low biomass before,during, and after the drought of 1980-1981. Fish species richness followed a cyclical pattern with peaks in 1975 and

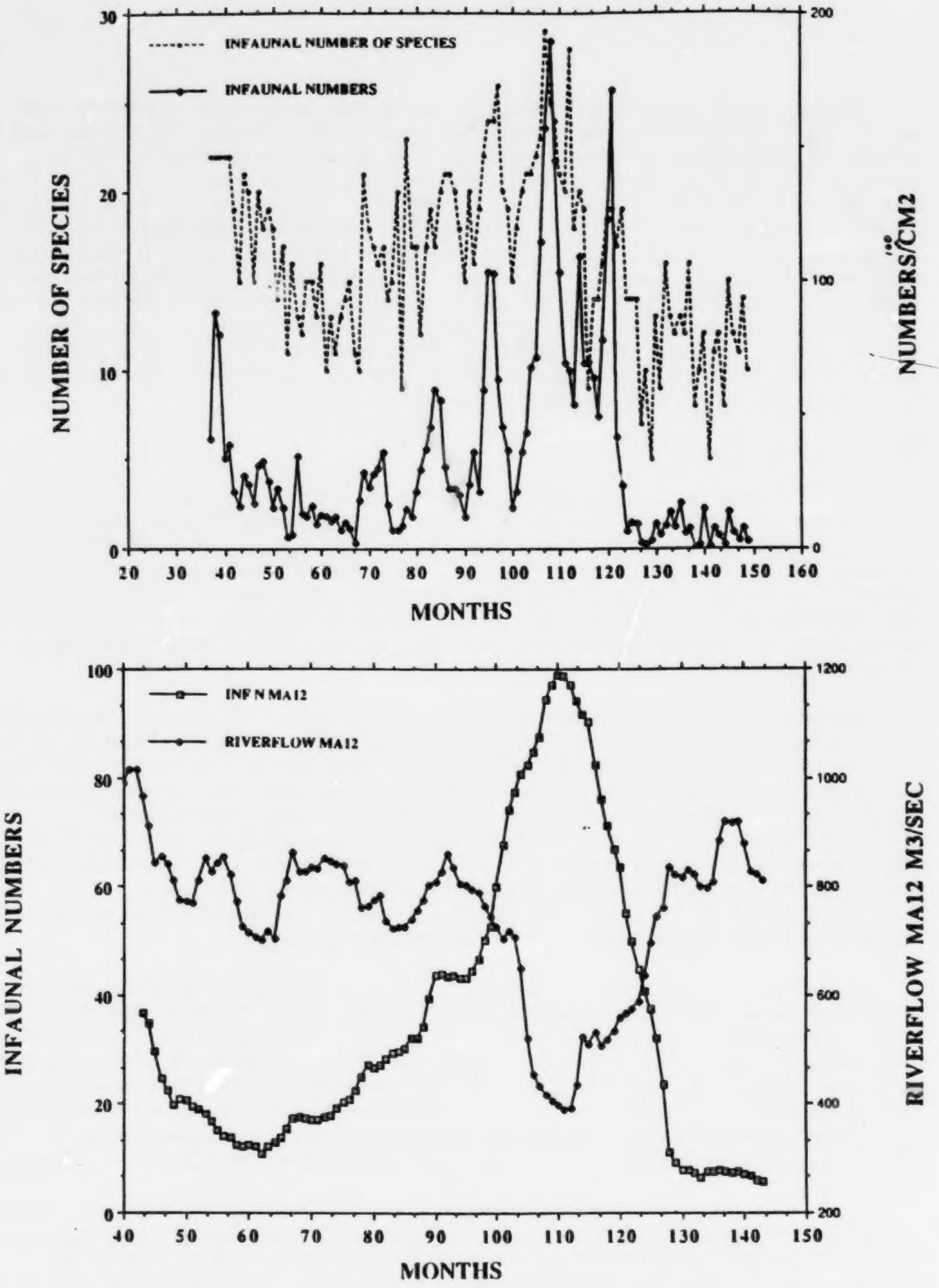

Figure 12. (a) Species richness and numerical abundance of infaunal macroinvertebrates in East Bay (all stations) taken monthly from 3/75-7/84. Data represent summed data. Peak levels occurred during the drought months and low levels were reached during the heat following the drought. (b) Moving averages of infaunal macroinvertebrate numerical abundance and species richness from 3/75-7/84.

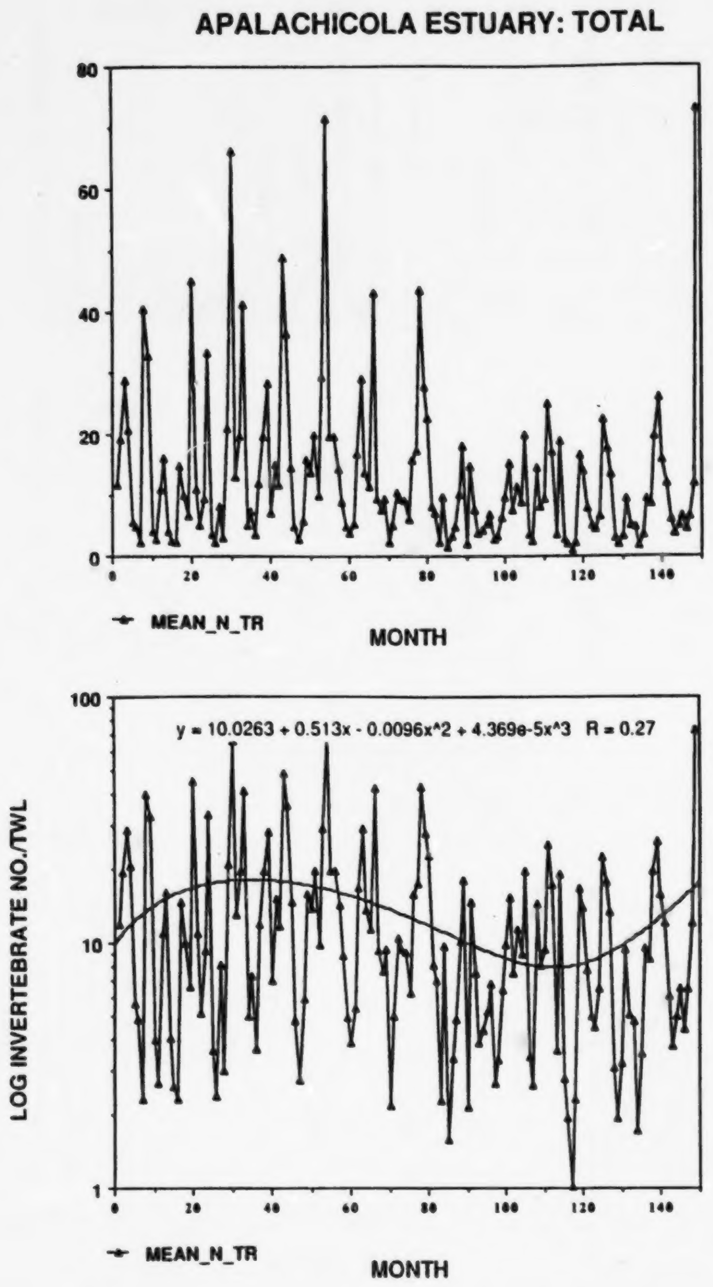

Figure 13. (a)Trawl-susceptible invertebrates (numbers, dry-weight biomass) taken monthly in the Apalachicola estuary (3/72-7/84). Monthly samples were summed. Log data were also analyzed using third order polynoial distribution showing a trend of abundance during the early years with low levels reached during the drought of the early 1980's. (b) Trawl-susceptible invertebrates (log-transformed) taken monthly in the Apalachicola estuary 3/72-7/84.

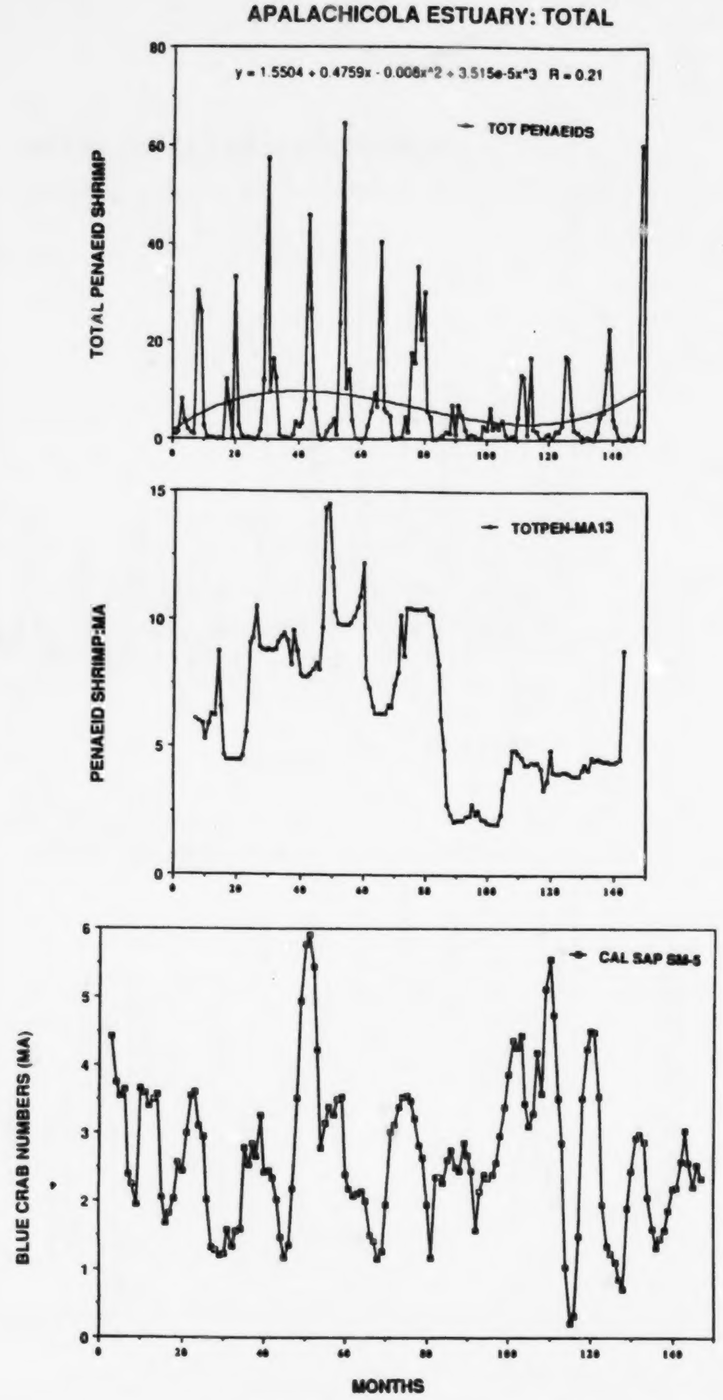

Figure 14. (a) Numbers of total penaeid shrimp taken monthly in the Apalachicola estuary (3/72-7/84). A third order polynomial distribution was used to show the trend of high numbers during the early years of the program. Also shown are moving averages of shrimp numbers (13-month) and moving averages of blue crab numbers (5-month) over the period of study. (b) Moving average of numbers of total penaeid shrimp taken in the Apalachicola estuary, 3/72-7/84. (c) Numbers of total blue crabs taken monthly in the Apalachicola estuary, 3/72-7/84.

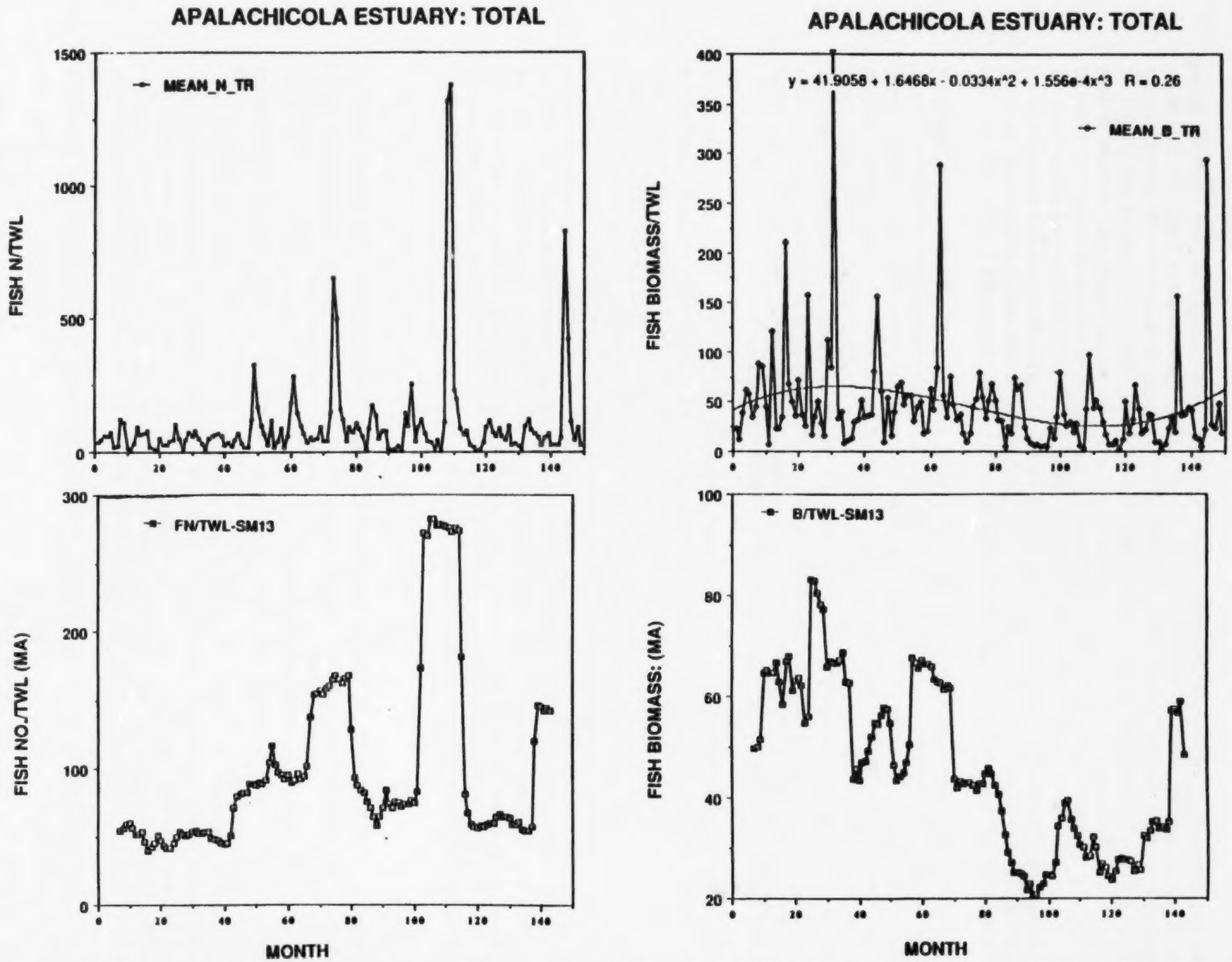

Figure 15. (a) Fish numbers taken monthly in the Apalachicola estuary from 3/72-7/84. Mean numbers summed over all stations were used. (b) Moving average of fish numbers taken monthly in the Apalachicola estuary, 3/72-7/84. (c) Total fish dry weight biomass taken monthly in the Apalachicola estuary, 3/72-7/84. (d) Moving average of total fish dry weight biomass taken monthly in the Apalachicola estuary, 3/72-7/84.

1984. These cycles tended to follow a very complex series of climatological changes and are currently under study. Low levels of species richness occurred during the drought. The statistical basis for the above relationships is given in Appendix IV. These results, while extremely complex, are consistent with known patterns of fish distribution in the Apalachicola system. Fish numbers are high during winter-early spring periods of high river flow, high POC, and organic carbon input from the river. By the summer-early fall months of high temperature and low dissolved oxygen, fish numbers decrease but biomass per trawl tow increases with seasonal growth and the arrival in the estuary of the larger predators. Fish species richness increases at this time. A multiple correlation matrix (with and without lags, Appendix IV) tends to corroborate this interpretation of the data. The various relationships shown with the lagged data indicate that a time series analysis may be more effective in delineating the cyclic associations of these variables. This is currently under study.

A review of the long-term climatological changes in the tri-river system and changes in the important fisheries of the Apalachicola estuary is given by Meeter et al. (1979). A spectral analysis of long-term trends of Apalachicola river flow and Georgia rainfall indicated cycles on the order of 6-7 years with possible longer term cycles that may reflect variations such as the low river flow during the mid-1950's. Cross-spectral analyses of these two variables showed that river flow and precipitation in Georgia are in phase. When commercial landings taken in the Apalachicola estuary are compared with the long-term trends of annual river flow, there were diverse results. Penaeid shrimp catches were not correlated ($p < 0.05$) with any of the other variables. Blue crab catches were positively correlated with river flow and negatively correlated with oyster catches. The oyster landings showed negative correlations with river flow and Apalachicola rainfall.

Such data reflect complex biological processes and socio-economic events which cannot be accounted for by use of the relatively simple statistics of correlation and regression. The considerable seasonal variations of the various commercial populations may mask the possible associations of such catches with longer term climatological changes and cycles. Optimal oyster growth occurs at salinities ranging from 15.0 to 22.5 ppt; during periods of high salinity, oysters are preyed upon by various stenohaline organisms that are usually excluded from the estuary by the low and varying salinities. However, extended periods of low salinity can have an adverse impact on oyster growth. Young blue crabs have a preference for low salinity, whereas the euryhaline white shrimp have no overt relationship to salinity. All such populations respond to complex combinations of variables such as river flow, temperature, salinity, productivity, predation, and other factors. There is little doubt that without the input of the Apalachicola River, these fisheries would largely disappear. However, the direct and indirect effects of changes in regional precipitation and river flow on the local fisheries are still not entirely clear owing in large part to the socio-economic variables that influence the commercial catch statistics.

Projections

A decrease in river flow of 24.2% (Hains' estimated GFDLx2 scenario) would cause direct losses of nutrients and POC that eventually would lead to proportionate losses in the detrital- and phytoplankton-based food webs of the estuary. A possible impact scenario for fishes is given in Figure 16, where the regressions determined during the 13-year field studies are applied to the Hains GFDLx2 scenario. Fish numbers were reduced in excess of 50% due to the loss of river flow according to this scenario. Once again, such projections are mere approximations owing to the problems associated with the statistical methods. If invertebrates are considered, after a brief increase of invertebrate numbers due to increased salinity and reduced predation, there would be an eventual reduction of infaunal and epibenthic macroinvertebrates with shifts in species representation as the estuary became less turbid and more saline. The reduction of food and increase of predators would lead to reductions of the oyster productivity; more oysters would be eaten by offshore stenohaline predators and there would be an increased toll due to oyster diseases that often accompany higher salinities. Such changes would probably be proportional to the change in river flow over the prolonged period of adjustment. Thus, the applications of the various scenarios of river flow changes could lead to proportional changes in the productivity of the estuary.

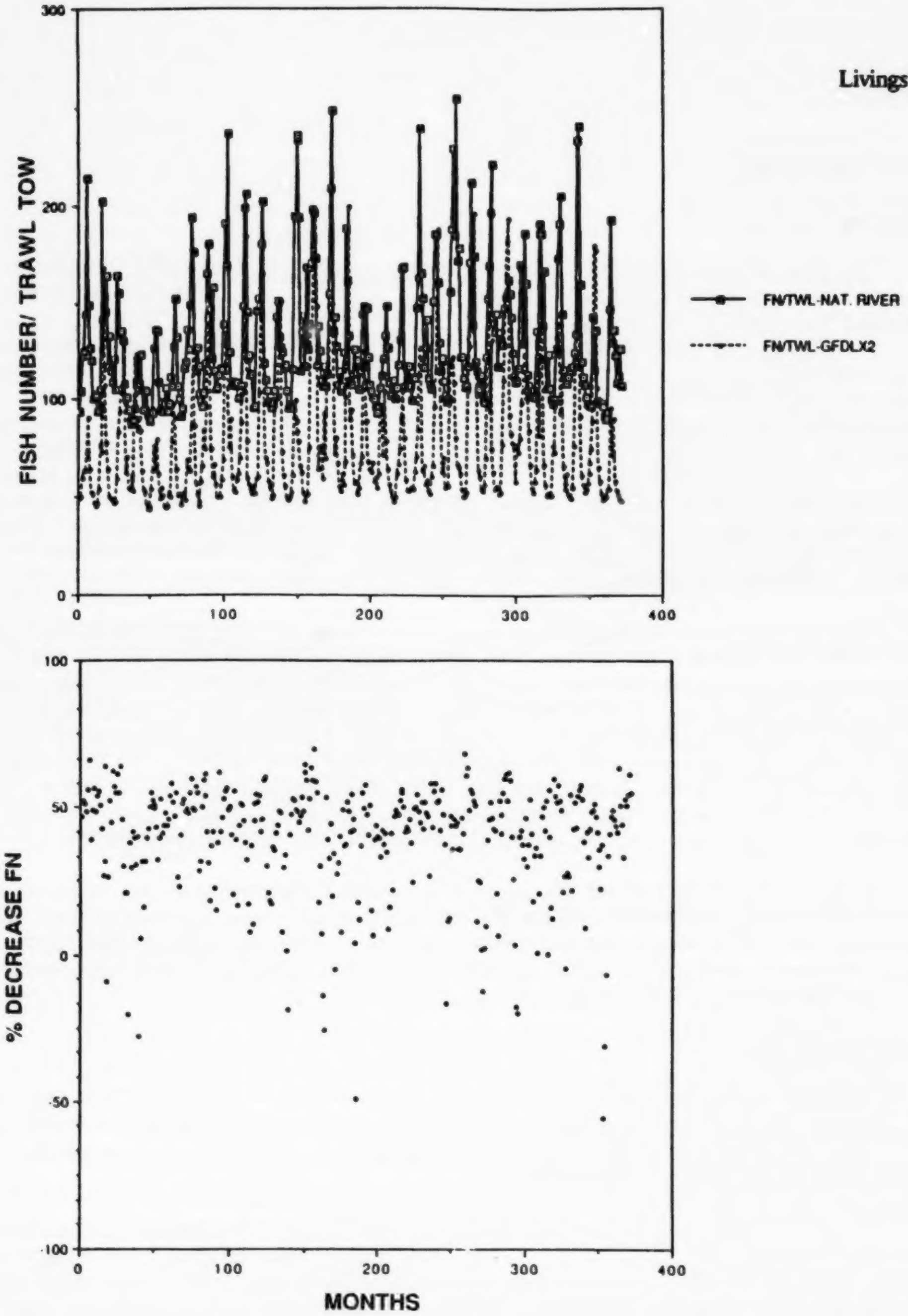

Figure 16. (a) Comparison of numbers of fishes taken per trawl under natural river flow conditions and numbers calculated from regressions of the GFDL river flow projections. Data are based on monthly averages of fishes taken from all stations in the Apalachicola estuary from 3/72-7/84. Also shown are the percent decrease of ratios of fish numbers taken under actual conditions and numbers produced from regression of GFDL projections of river flow. (b) Comparison of numbers of fishes taken per trawl under natural river flow conditions and numbers calculated from regressions on the GFDL river flow projections. Figure is presented as the percent decrease of fish numbers.

TEMPERATURE

Scenarios

The EPA scenarios (GISS, GFDL) were run with ambient data taken within the tri-river drainage system. Comparisons were made during the 13-year period of study (3/72-7/84). Previous studies (Livingston, 1984b) indicated that monthly average air temperatures in Apalachicola, Florida (NOAA station) are correlated ($r = 0.90$, $p < 0.00001$) with water temperatures in the shallow estuaries along the north Florida gulf coast. The use of air temperature as a predictor of estuarine water temperature in the Apalachicola Bay system is thus valid. However, unlike river flow and salinity, no regressions were run with the biological data. The temperature changes would affect the bay thoughout any given year; however, the summer monthly temperatures would reach lethal levels which differ from species to species. This is a threshhold effect that is difficult to quantify by the use of correlations and regressions. Instead, the projected changes in temperature were calculated according to the transformation of existing data by use of the various models. A literature search gave the various temperature tolerances of the important estuarine species in the Apalachicola estuary. From this analysis, a frequency diagram was constructed giving the projected changes in the estuary based on the temperature tolerances of indigenous species.

The ambient (bay-wide mean) water temperature trends over the period of study (Figure 17) indicated that winter low temperatures showed progressive declines from 1975 to 1977 at which time the lowest temperature levels of the period were observed. From 1977 to 1980, there was a progressive increase in low winter temperature levels. A time-series analysis of air temperatures in Apalachicola from 1930 to 1979 (Livingston, 1984b) showed that the lowest monthly minimum air temperatures occurred during winter months in 1940, 1948, 1958, 1968, and 1977. The approximate period of such low winter temperatures appears to be within a range of 8-10 years. Summer high mean temperatures varied less widely with a range approximating 28-31°C. The highest summer mean temperatures in the Apalachicola system occurred in 1972, 1976, 1977, 1980, and 1983. No long-term patterns of recurring high summer temperatures were evident.

A comparison of the ambient temperature vs the GISS transformation showed relatively little difference in the summer high temperatures (Figure 18); winter lows would increase somewhat more in terms of absolute increases. The GFDL model, however, showed substantial increases in the summer highs. Such differences range between 5 and 9°C (Figure 18). According to the GFDL model, summer high average temperatures in the Apalachicola estuary would range between 15 and 16°C.

Biological Response

Thermal stress operates at both ends of the temperature spectrum. However, because the high temperatures (summer) along the upper gulf coast approach the upper lethal levels of various estuarine species, and because the GISS and GFDL projections indicate general increases in such temperature levels, the upper lethal levels will be emphasized in this review.

There is a considerable literature concerning the effects of temperature on poikilothermic (aquatic) organisms (Collins, 1985; Darcy, 1985; Lassuy, 1983; Muncy, 1984; Reagan ,1985; Reagan and Wingo, 1985; Stanley and Sellers, 1986; Sutter and McIlwain, 1987;Van Den Avyle, 1984). Brett (1970) and Warren (1971) have reviewed various aspects of the effects of temperature on fishes and invertebrates. Temperature is a major environmental determinant of the distribution of aquatic organisms in space and time. Factors such as acclimatization and adaptation complicate the interpretation of direct effects of temperature on the distribution of aquatic organisms. Temperature tolerance is genetically controlled with modifying features including salinity, photoperiod, season, size, diet, reproductive state, and the acclimatization history of the population in question. The rate of temperature change along with the exact nature of the state variable in question (e.g., respiration, behavior, reproduction, mortality) and the thermal history of the subject all interact to determine the mechanism

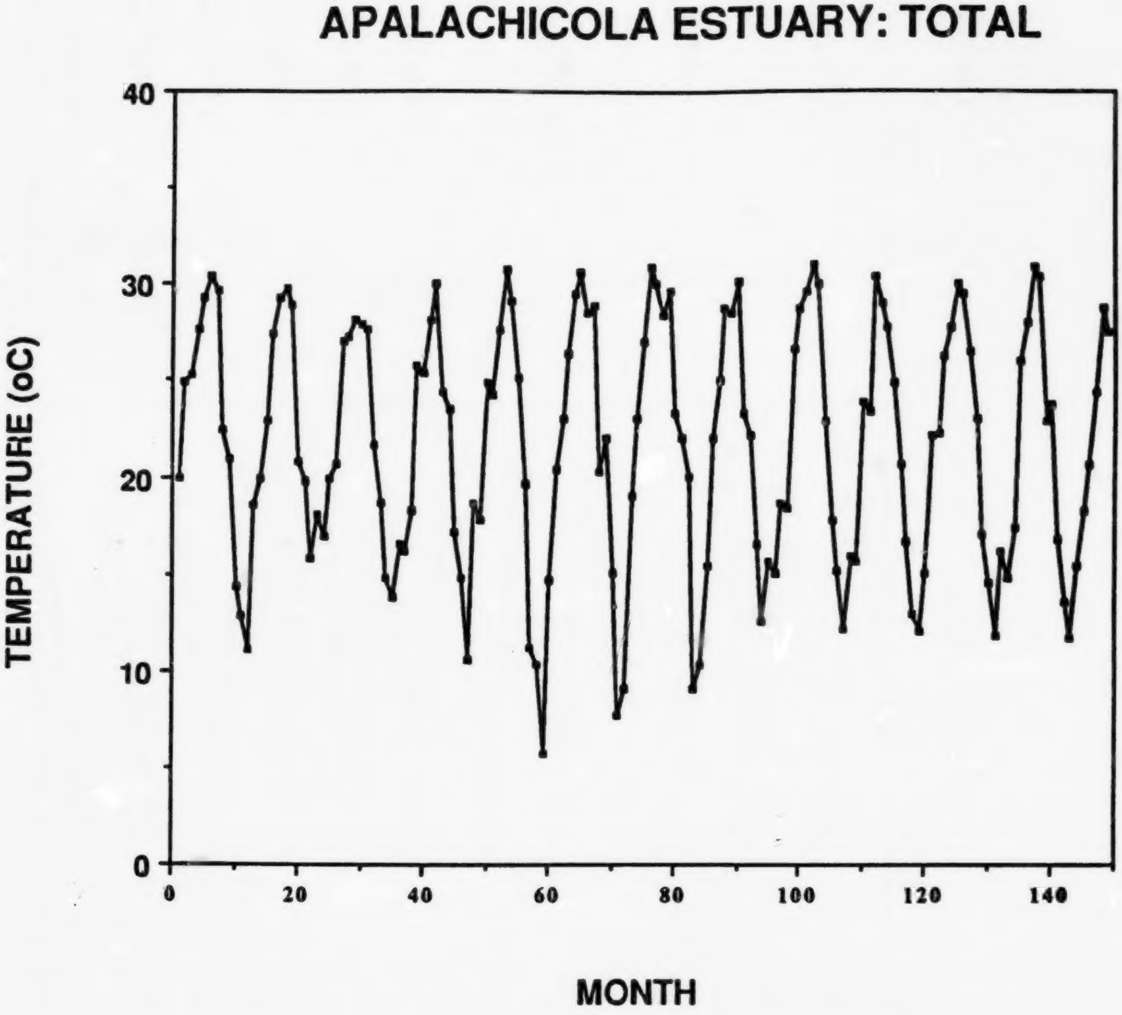

Figure 17. Monthly temperature changes in the Apalachicola estuary from 3/72-7/84. Numbers are based on mean values taken from all stations (surface and bottom) in the Apalachicola estuary.

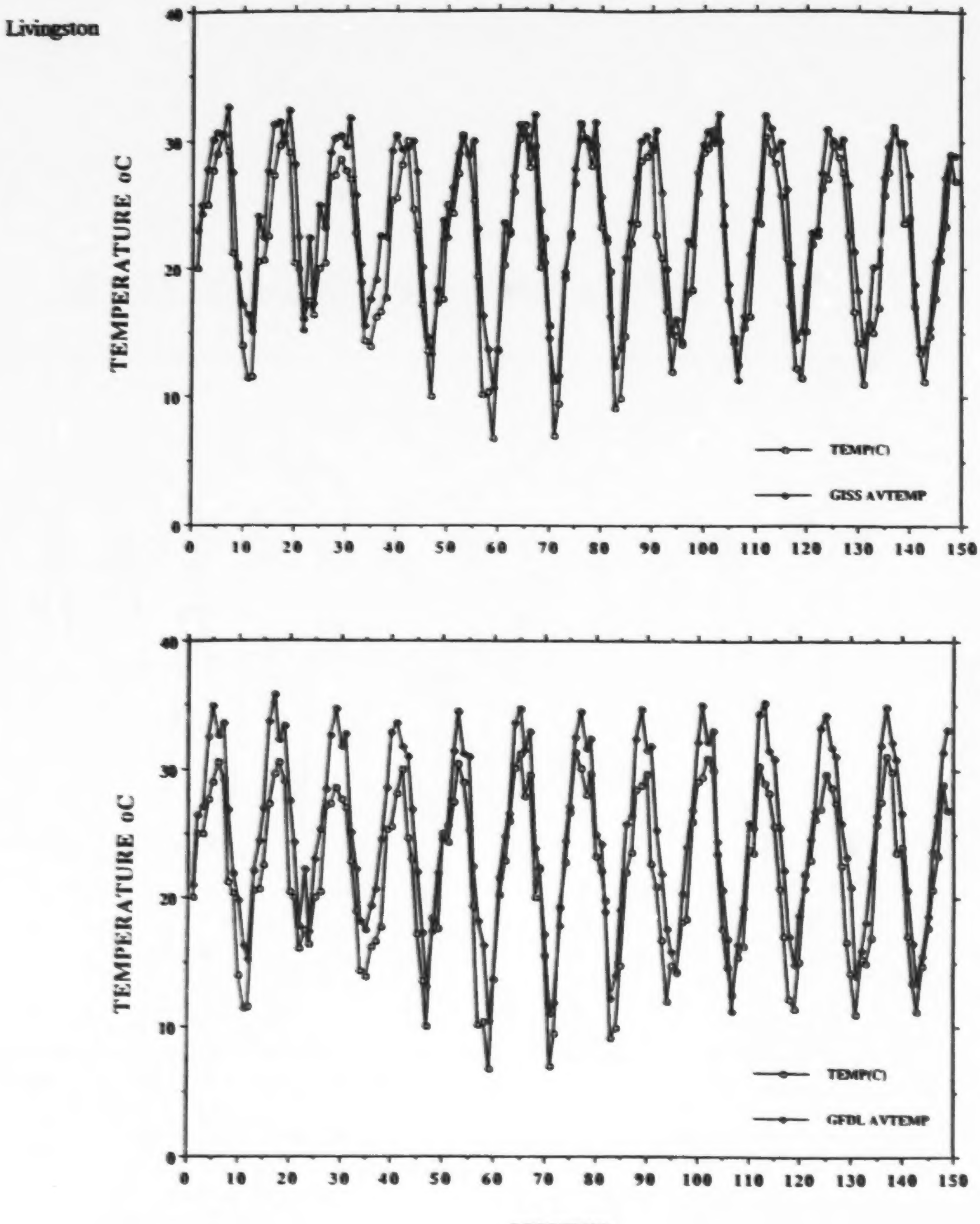

Figure 18. Comparisons of ambient water temperature and (a) GISS transformations and (b) GFDL transformations (projections) of water temperature in the Apalachicola estuary (3/72-7/84). Data are based on monthly means of all stations.

of thermal tolerance in a given population. Indirect effects of associated variables such as dissolved oxygen (e.g., temperature is a primary determinant of the level of dissolved oxygen in water) complicate the direct and indirect effects of temperature on aquatic organisms.

A review was made concerning the upper thermal collection limits (e.g., the highest temperature at which an aquatic species was taken in the field) and the upper critical or incipient lethal temperature (e.g., that temperature level which will eventually cause the death of a stated fraction of the test organisms, usually 50%) (Table 5). There are species-specific trends with organisms such as the lutjanids having relatively low upper thermal collection limits. Important species in terms of abundance and biomass in the Apalachicola estuary include oysters (upper thermal collection limit or U.T.C. of 36°C), blue crabs (upper thermal tolerance or U.T.T. = 30-33°C), penaeid shrimp (U.T.C. = 35-35.5°C), stone crabs (U.T.C. = 32°C), seatrouts (U.T.C. = 34.9-35°C), catfishes (U.T.C. = 34.9-37°C), mullet (U.T.C. = 34.5°C), and southern flounder (U.T.C. = 35°C). Between the temperatures of 30-37.5°C (usually not exceeding 35°C), there could be a progressive reduction of the dominant estuarine species found during summer periods in the Apalachicola Bay system.

An analysis was made of the major dominant species in the Apalachicola estuary with particular emphasis on commercially important species (Tables 6 and 7). According to the GFDL projections of summer high temperatures, blue crab larvae and juveniles, spotted seatrout, oyster larvae, pinfish, flounder, and croaker would be virtually eliminated. Severe impacts would occur with the redfish population. In the GISS model projections, only blue crab larvae and juveniles would be eliminated, although spotted seatrout, oyster larvae, pinfish, and flounder would be severely affected. Under current (ambient) conditions, only blue crab larvae have been adversely affected by high summer temperatures; since larvae move into the bay mostly during winter periods, such an impact would be hardly felt. In any case, the GFDL and GISS projections of temperature increases would have a major adverse impact on the estuary.

The general increase of winter low temperatures would also have a profound influence on the species composition of aquatic organisms in the Apalachicola estuary. The biota would probably become more tropical in terms of temperature-tolerant species with general increases in species richness and diversity and general reductions of overall fish and invertebrate production in terms of dry-weight biomass. Such projections would be strongly influenced by potential (co-existing) changes in other variables such as salinity and dissolved oxygen. According to Yanez-Arancibia et al. (1985), tropical estuaries in the Mexican portion of the Gulf of Mexico have the highest phytoplankton productivity in areas receiving high levels of river flow. Thus, the overall production of the Apalachicola estuary would depend to a considerable degree on changes in the mass flow of freshwater into the estuary regardless of the ambient temperature of the system. Commercial fish catches in Mexican estuaries are directly dependent on the river discharge. Coastal wetlands are also an important component in such productivity (Yanez-Arancibia et al., 1985).

Overall, an average increase of 4-5°C in the Apalachicola estuary would probably cause a major disruption of the aquatic fauna at both ends of the temperature spectrum. It is doubtful that there would be enough time for species tolerant to the high temperature and accompanying low dissolved oxygen to adapt to such conditions, leaving only the organisms already tolerant to the projected temperature increases. When other more restrictive functions such as growth and reproduction are taken into consideration, the impact of the projected temperature increases in the Apalachicola region become even more meaningful since few organisms are adapted to maintain their population numbers at temperatures higher than 35-36°C. The major break point in the Apalachicola estuary would be sustained summer maximum temperatures of 35°C and higher.

The question of acclimatization is difficult to address. Temperature increases are projected to occur over a relatively short time period. There could be some adaptive response of existing species, although this is unlikely since there is some evidence (Livingston, unpublished data) that many of these species are already temperature limited. This is not a matter of succumbing to a single factor, since the effects of such temperature changes would result in habitat alterations that would simply render the estuary unfit for certain species. Another complicating factor would be the synergistic effects of temperature and other state variables such as

Table 5. Thermal Tolerance Levels of Selected Estuarine Species*

Reference) Species	Upper Thermal Collection Limit	Upper Thermal Tolerance (Upper Critical or Incipient Lethal Temperature)	
INVERTEBRATES			
1) *Callinectes sapidus*		30°C (larvae) 33°C (juveniles)	
2) *Menippe mercenaria*	32°C		
3) *Palaemonetes pugio*	38°C	38°C 35°C (larvae)	
4) *Penaeus duorarum*	35.5°C		
5) *Penaeus setiferus*		42°C	
6) *Penaeus aztecus*	35°C		
7) *Crassostrea virginica*	36°C	48°C >35°C inhibits larval growth	
FISH			
8) *Anchoa mitchelli*	39.8°C		
8) *Anchoa hepsetus*	34.9°C		
9) *Archosargus probatocephalus*	35.1°C		
10) *Arius felis*	37°C	near 39°C	
10) *Bagre marinus*	>34.9°C		
11) *Brevoortia patronus*	34.9°C		
12) *Cynoscion arenarius*	35°C		
13) *Cynoscion nebulosus*	34.9°C		
12) *Cynoscion nothus*	30°C		
14) *Fundulus heteroclitus*	40-42°C		
14) *Fundulus majalis*	40-42°C		
15) *Lagodon rhomboides*	33-34°C	35-36°C avoided	
16) *Lutjanus analis*	27.8°C		
16) *Lutjanus griseus*	32.5°C		
16) *Lutjanus synagris*	28.9°C		
17) *Micropogonias undulatus*		35.5°C 38°C (juveniles)	36°C
18) *Mugil cephalus*	34.5°C	41.3°C (juveniles)	
16) *Ocyurus chrysurus*		33.5-34°C	
19) *Orthopristes chrysoptera*	36°C		
20) *Paralichtys lethostigmata*	35°C		
21) *Pogonias cromis*	33°C		
22) *Sciaenops ocellatus*	37.5°C		

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* Data are given as the Upper Thermal Collection Limits and as the Upper Thermal Tolerances. Refernces and significance of the numbers are given in the text.

Table 6. Distribution of Upper Thermal Tolerances of Dominant Species in the Apalachicola Estuary from 1972-1977. Temperatures that Eliminate Species from the Bay are Listed Along With Projections that Would be Eliminated According to Different Scenarios During the Summer Months from 1972-1977 (Months Start in June of Each Year)

	AVERAGE MAXIMUM MONTHLY TEMPERATURE					
DATE summer	°C GFDL	ELIMINATE	°C GISS	ELIMINATE	°C ACTUAL	ELIMINATE
1972	37.3	bcl.bcj,ol st,p,c,f	34.8	bcl.bcj	30.9	bcl
	39.6	bcl.bcj,ol st,p,c,r,f	35.2	bcl,bcj,ol st,p,f	32.1	bcl
	37.6	bcl,bcj,ol st,p,c,r,f	35.3	bcl.bcj,ol st,p,f	33.1	bcl,bcj
	38.1	bcl,bcj,ol st, p,c,r,f	37.1	bcl,bcj,ol st,p,c,f	31.9	bcl
1973	37.7	bcl,bcj,ol st,p,c,r,f	35.2	bcl,bcj,ol st,p,f	31.3	bcl
	40.6	bcl,bcj,ol st,p,c,r,f	36.2	bcl,bcj,ol st,p,c,f	33.1	bcl.bcj
	36.2	bcl,bcj,ol st,p,c,f	33.9	bcl,bcj,	31.7	bcl
	37.6	bcl,bcj,ol st,p,c,r,f	36.6	bcl,bcj,ol st,p,c,f	31.3	bcl
1974	36.5	bcl,bcj,ol st,p,c,f	33.4	bcl,bcj	28.9	
	40.1	bcl,bcj,ol st,p,c,r,f	34.1	bcl,bcj	33.1	bcl,bcj
	36.4	bcl,bcj,ol st,p,c,f	35.7	bcl,bcj,ol st,p,f	32.6	bcl
1975	38.0	bcl,bcj,ol st,p,c,r,f	34.1	bcl,bcj	30.3	bcl
	37.9	bcl,bcj,ol st,p,c,r,f	33.2	bcl,bcj	32.5	bcl
	36.7	bcl,bcj,ol st,p,c,f	35.5	bcl,bcj,ol st,p,f	28.9	
1976	36.6	bcl,bcj,ol st,p,c,f	34.1	bcl.bcj	30.3	bcl
	40.0	bcl,bcj,ol st,p,c,r,f	35.6	bcl,bcj,ol st,p,f	32.5	bcl
	36.4	bcl,bcj,ol st,p,c,f	34.1	bcl,bcj	31.9	bcl
	35.1	bcl,bcj,ol st,p,f	34.1	bcl,bcj	28.9	
1977	33.0	bcl,bcj	35.2	bcl,bcj,ol st,p,f	30.7	bcl
	34.6	bcl,bcj	38.4	bcl,bcj,ol st,p,r,f	34.5	bcl,bcj
	40.9	bcl,bcj,ol st,p,c,r,f	38.1	bcl,bcj,ol st,p,c,r,f	35.0	bcl,bcj,ol p,f
	42.5	bcl,bcj,ol ws,st,p,c m,r,f	35.2	bcl,bcj,ol st,p,f	33.0	bcl.bcj

SPECIES	UPPER THERMAL TOLERANCE (°C)
blue crab larvae (bcl)	30
blue crab juveniles (bcj)	33
white shrimp (ws)	42
oyster larvae (ol)	35
spotted seatrout(st)	34.9
pinfish(p)	35
croaker (c)	36
mullet (m)	41.3
redfish (r)	37.5
flounder (f)	35

(TEMPERATURES THAT ELIMINATE
SPECIES FROM BAY)

Table 7. **Elimination of Species (%) Based on Projected Increases in Temperature According to a Comparison of the GFDL and GISS Scenarios With Ambient Conditions in the Apalachicola Estuary (Summer Months: 1972-1977)**

	gfdl	giss	ambient
blue crab larvae.........	22	22	17
blue crab juveniles...	22	22	6
spotted seatrout........	20	13	1
oyster larvae...............	20	13	1
pinfish............................	20	13	1
flounder..........................	20	13	1
croaker............................	19	4	0
redfish.............................	12	2	0
mullet...............................	1	0	0
white shrimp................	1	0	0

salinity. Since we know little about how the system would respond to each of the projected changes as sole entities, projections of interacting effects are even more problematical. The interactions of such changes on single populations should also be related to altered levels and timing of nutrient input and primary productivity in the Apalachicola estuary.

According to Yanez-Arancibia et al. (1985), the temperature regime of the Terminos Lagoon, a tropical estuary on the Mexican Gulf of Mexico, approximates that of the Apalachicola estuary with high temperatures around 30-31°C. The projected estuarine temperatures of the future Apalachicola system were predicated on an equivalence with increased air temperature. It is possible that conditions in the Gulf of Mexico could alter such a situation whereby the water temperature would be cooler than the air temperature. Such equivalence should thus be acknowledged as an untested assumption subject to further review. In any case, such studies of tropical estuaries indicate that these systems are highly productive in terms of fisheries output and are dependent on factors (e.g., freshwater input, wetlands vegetation) similar to the temperate estuaries to the north.

SEA LEVEL CHANGES

Projected Effects

One complicating feature of the projected impact of climatic changes on the Apalachicola system includes sea level alterations. A series of projected changes have been developed by Dr. Richard Park (Table 8). Such changes should be placed in the perspective given by the various sources of POC for the Apalachicola estuary as summarized in Table 9 (Livingston, 1984a). The Apalachicola Bay system receives input in terms of organic carbon and nutrients from various portions of the drainage system. The river flooding during winter-early spring periods has been associated with input of POC and nutrients. Phytoplankton blooms during the spring and fall months combine with input from the seagrass beds and fringing salt water marshes during the summer and fall months to provide the bay with POC and dissolved and particulate forms of nutrients. Such inputs are thought to be mediated by the activities of microorganisms, which transform the dissolved components into the enriched (e.g., N, P) particulate organic matter that forms the basis of the detrital food webs of the estuary. There is a certain equivalence of input from the freshwater wetlands, the coastal marshes, and the seagrass beds. Phytoplankton productivity accounts for over 200% of the total of the above sources. The river has been associated with such production along with the movement of nutrients and POC within the river-bay system.

The models provided by Dr. Richard Park (Butler University, Indianapolis, Indiana) will serve as the basis of this section of the report. No quantitative (statistical) projections can be made with such data, since there is little precedence for the major loss of wetlands in this region. However, there are enough data concerning the function of such areas that impacts can be evaluated concerning losses of major portions of the Apalachicola wetlands.

Four scenarios of sea level change are presented (Tables 8 and 9): baseline projected changes (no change in current rate of sea level increase); sea level increase of 0.5 m by 2100; sea level increase of 1.0 m by 2100; and sea level increase of 2.0 m by 2100). The projected increases in sea level by the year 2100 (File FLAAPLAC; Scenario 1) would be associated with some decrease in swamp habitat; high marshes would be hard hit and low marshes would also suffer some losses. No significant changes would occur in the freshwater marshes or the overall water surface.

The second scenario (0.5-meter increase by 2100) would be associated with more significant changes. About 1/3 of the swamps and nearly 1/5 of the freshwater marshes would be gone along with nearly all the low and high marshes around the bay. The increase of mangrove swamps is difficult to factor into these evaluations; if mangroves would eventually grow in the Apalachicola region, there would be some amelioration of the effects owing to the projected losses of the existing wetlands. However, due to the uncertainty involved in this projection, the mangroves will not be taken into account in this analysis.

Table 8. Projected Changes in the Apalachicola Region Due to Sea Level Rises (Based on Data Provided by Dr. Richard Park: Butler University, Indianapolis, Indiana)

I. File FLAAPLAC

A. Scenario 1:

Time period: 1987-2100
Sea level rise: 1.2 mm/year (baseline).

Projected changes:

1. Swamps: 9.4616 to 6.7053: decrease of 2.7563= 29.1% decrease.
2. Freshwater marshes: 1.4595 to 1.2708; decrease of 0.1887=12.9% decrease.
3. High marsh: 1.1858 to 0.3656: decrease of 0.8202 = 69.1% decrease.
4. Low marsh: 3.4213 to 2.3268: decrease of 1.0945 = 32.0% decrease.
5. Mangrove: no change.
6. Water: 77.4019 to 81.5257: increase of 4.1238 = 5.3% increase.

B. Scenario 2:

Time period: 1987-2100.
Sea level rise: 0.5 m increase.

Projected changes:

1. Swamps: 9.456 to 6.2579; decrease of 3.1981; = 33.8% decrease.
2. Freshwater marshes: 1.4595 to 1.1715 = decrease of 0.2880= 19.7% decrease.
3. High marsh: 1.1858 too.0353; decrease of 1.1505 = 97.0% decrease.
4. Low marsh: 3.4213 to 0.3879 = decrease of 3.0334 = 88.9% decrease.
5. Mangrove: 0 to 3.0575 = increase of 3.0575.
6. Water: 77.4019 to 81.857 = increase of 4.4551 = 5.8% increase.

Table 8. (continued)

C. Scenario 3:

Time period: 1987-2100.
Sea level rise: 1 meter increase.

Projected changes:

1. Swamps: 9.456 to 5.4695 = decrease of 3.9865 = 42.2% decrease.
2. Freshwater marshes:1.4595 to 0.999 = decrease of .4215 = 28.9% decrease.
3. High marsh: 1.1858 to 0.0350 = decrease of 1.1508 = 97.0% decrease.
4. Low marsh: 3.4213 to 0.0616= decrease of 3.397 = 98.4% decrease.
5. Mangrove: 0 to 2.1305= increase of 2.1305.
6. Salt marsh: 4.6071 to 0.0966= decrease of 4.5105= 97.9% decrease.
7. Water: 77.4019 to 85.2671; increase of 7.8652 = 10.2 % increase.

D. Scenario 4:

Time period: 1987-2100.
Sea level rise: 0.0756 to 2.004 = 2 meter increase.

Projected changes:

1. Swamps: 9.456 to 4.1158: decrease of 5.3402 = 56.4% decrease.
2. Freshwater marshes: 1.4595 to 0.2461: decrease of 1.2134 = 83.4% decrease.
3. High marsh: 1.1858 to 0.0199: decrease of 1.1659 = 98.3% decrease.
4. Low marsh: 3.4213 to 0.0322: decrease of3.3891 = 99.0% decrease.
5. Mangrove = 0 tp 1.8006 = increase pf 1.8006.
6. Salt marsh: 4.6071 to .0521: decrease of 4.555 = 98.9% decrease.
7. Water: 77.4019 to 88.3808: increase of 10.9789 = 14.1% increase.

Table 9. Compendium of Data Concerning Different Terrestrial and Aquatic Habitat/Productivity Variables in the Apalachicola Drainage Basin

Distribution and area of major bodies of water along the coast of Franklin County (north Florida) with areas of oysters, grassbeds, and contiguous marshes.

Water body	Area (ha)	Oysters (ha)	Grassbeds (ha)	Marshes (ha)
St. Vincent Sound	5,539.6	1,096.5	---	1,806.9
Bay	20,959.8	1,658.5	1,124.7	703.4
East Bay	3,980.6	66.6	1,433.5	4,606.1
St. George Sound (West)	14,746.8	1,488.8	624.3	751.9
St. George Sound (East)	16,015.5	2.6	2,767.3	810.8
Alligator Harbor	1,637.0	36.7	261.3	144.3
Total	62,879.3	4,349.7	6,211.1	8,850.4
Percent of total water area	100	7	10	14

Terrestrial habitats and land-use patterns in the immediate watershed of the Apalachicola Bay system (Florida Bureau of Land and Water Management 1977).

Category	Total area (ha)	% of total
Residential	2,461	1.3
Commercial, services	178	0.1
Transportation, utilities	218	0.1
Mixed urban or built-up areas	27	0.0
Other urban or built-up areas	39	0.0
All urban or built-up areas	2,922	1.5
Cropland and pasture	78	0.0
Other agriculture	4	0.0
All agricultural land	82	0.0
Herbaceous rangeland	13	0.0
Rangeland	13	0.0
Evergreen forest land	68,598	35.7
Mixed forest land	36,396	18.9
All forest land	104,994	54.6
Streams and canals	1,469	0.8
Lakes	452	0.2
Reservoirs	10	0.0
Bays and estuaries	62,879	24.3
All water	64,810	25.4
Forested wetland	25,562	13.3
Nonforested wetland	8,465	4.4
All wetlands	34,027	17.7
Beaches	1,441	0.7
Quarries and pits	25	0.0
Transitional areas	110	0.1
All barren land	1,575	0.8
Total area of Franklin County:	198,398	

Table 9. (continued)

	Apalachicola estuary		Apalachicola Bay system		
Vegetation	Net in situ productivity mt C yr^{-1}	Net input mt C yr^{-1}	Net in situ productivity mt C yr^{-1}	Net input mt C yr^{-1}	Season of maximum input
Freshwater wetlands	360,000	30,000	360,000	30,000	winter/spring
Coastal marshes	37,714	37,714(?)	46,905	46,905(?)	late summer, fall(?)
Phyto-plankton	103,080	103,080	233,284(?)	233,284(?)	spring and fall
Seagrass beds	8,953	8,953	27,213	27,213	summer-fall

	Acres					
Mapping category	Upper river	Middle river	Lower river from Wewahitchka to Sumatra	Lower river from Sumatra to mile 10	Lower river from mile 10 to mouth	Total
Pine	136	672	0	204	0	1,010
Sweetgum-Sugarberry-Water oak-Loblolly Pine	642	1,440	154	474	0	2,710
Water hickory-Sweetgum-Overcup oak-Green ash-Sugarberry	12,500	32,200	15,800	1,770	48.0	62,300
Tupelo-cypress with mixed hardwoods	1,170	1,860	8,310	15,800	6,920	34,100
Tupelo-cypress	2,420	2,270	6,240	10,300	456	21,700
Pioneer	0	150	19.2	0	0	169
Marsh	0	0	0	0	9,030	9,030
Open water	2,730	3,110	1,540	2,010	1,260	10,700
Unidentified	1,020	748	81.3	76.8	19.2	1,950
Total	20,600	42,500	32,100	30,600	17,700	144,000

		Annual output minus input (metric tons)			Areal yield (g m^{-2} yr^{-1})		
Drainage basin	Area (km^2)	Carbon	Nitrogen	Phos-phorus	Carbon	Nitrogen	Phos-phorus
Apalachicola-Chattahoochee-Flint	50,800	213,800	21,480	1,652	4	0.4	0.03
Chattahoochee-Flint	44,600	142,700	17,860	1.340	3	0.4	0.03
Apalachicola-Chipola	6,200	71,100	3,620	312	12	0.6	0.05
Apalachicola	3,100	41,500	1,060	237	13	0.3	0.08
Chipola	3,100	29,600	2,560	75	10	0.8	0.02

A one meter increase in the sea level would be accompanied by further reduction of the freshwater marshes (28.9% decrease). The rest of the marsh systems would be virtually gone with the exception of the mangrove areas. The 2-meter increase by the year 2100 would be associated with the loss of virtually all the wetlands systems (freshwater, brackish, and marine) according to the model projections.

An estimate of how such progressive losses of the wetlands in the Apalachicola region would translate into a loss of POC to the system is given in Table 10. It should be acknowledged that these calculations are based on major assumptions that cannot be effectively evaluated in terms of their impact on the final results. With the exception of the mangroves, the model assumes no replacement of marshes as the sea level rises. The exact relationship of the marshes to the seagrass beds remains undetermined; it is possible that an adverse impact on the marshes and wetland systems associated with the river-estuary could adversely affect the SAV in the estuary, since such plants may be dependent on nutrients from such areas. The SAV, on the other hand, would have more area to grow as the water surface increased; hence the direct increase of SAV as the water areas expanded with time (Table 10). The direct loss of wetlands may have an equivalent effect on phytoplankton productivity which may, in turn, be inversely associated with the SAV production (inversely due to competition for nutrients and light). The projected changes in the carbon regime of the bay are also complicated by unknown effects on the microbiota: microbial activity is critical to any evaluation of POC input to the estuary. The extremely complex relationships of such elements have not been determined. However, for the sake of simplicity, I related the direct loss of phytoplankton productivity with an average of the loss of the freshwater marshes and the high/low marshes of the system. There is little doubt that the nutrients from such systems, as mediated by freshwater flow into the estuary, are an important component in the high levels of phytoplankton productivity in the bay system.

Based on the above assumptions, the calculations (Table 10) would indicate a considerable loss of POC to the estuary by the year 2100. The salt water marshes would disappear quickly. The phytoplankton losses would be considerable as the surrounding wetlands disappeared. Even the baseline decreases would be associated with about one-third of the POC of the system. The projected losses from the 0.5- to 2.0-m increases in sea level would be considerable. The increase in SAV over this period would not appear to make up for such losses. Overall, the input of POC due to the primary allochthonous and autochthonous sources would go from 337,385 metric tons annually in 1987 to 51,955 metric tons in 2100. This loss could then be projected to those elements of the estuary that are directly involved with the detrital foodwebs. Since most of the major fisheries are tied in some way to the detrital food webs of the estuary, the projected loss could be applied to similar losses of the shrimp, oyster, and finfish production of the estuary.

The projected changes in the upper regions of the tri-river basin are equivalent to those treated above and would not appreciably change the estimates of carbon loss by the year 2100. The speculation concerning the appearance of mangroves in the Apalachicola region was not addressed here and can be considered to be a neutral issue by the year 2100 owing to the eventual reduction and loss of such vegetation.

Table 10. Projected Changes of the Net Input of Organic Carbon (mtons/year) to the Apalachicola Bay system (Livingston, 1984a) According to Various Scenarios of Sea Level Rise (Table 8). (Seagrass Changes are Related to Increases in Water Areas; Phytoplankton Changes are Related to Changes in the Average Decreases of Freshwater Marshes, High Marsh Areas, and Low Marsh Areas)

FACTOR	FRESH WETLANDS	SEA-GRASS	SALT MARSHES	PHYTO-PLANKTON	TOTAL
Scenario (1987-2100)					
CURRENT	30,000	27,200	46,905	233,280	337,385
BASELINE INCREASES (1.2MM/yr)	26,100	28,700	23,500	144,640	222,940
0.5 M INCREASE	24,000	28,800	4,690	71,450	128,940
1.0 M INCREASE	21,300	30,100	940	58,790	111,130
2.0 M INCREASE	4,980	31,035	780	15,160	51,955

CHAPTER 4

DISCUSSION

RIVER FLOW PROJECTIONS AND EFFECTS

The GISS model of precipitation changes predicted an average increase of 2 cm of rainfall per month, whereas the GFDL model predicted an average decrease of 2 cm per month when applied to the Columbus, Georgia, precipitation records over the period of study (3/72-7/84). However, the Hains model (GFDLx2) projected a reduction of over 25% of the river flow. Such an alteration could be associated with proportional losses of nutrients and POC to the estuary in addition to changes in the salinity regime that could cause shifts in the biotic relationships and losses of some part of the fisheries potential of the system.

The Apalachicola River delivers considerable quantities of particulate organic matter and nutrients to the estuary. Such substances are associated with the various food webs in the estuary, especially during the winter-early spring peak flows. Phytoplankton production during the warmer periods is thought to be related to nutrients deposited in the bay during various times of the year. The methods used to evaluate changes in flow preclude an exact measure of the seasonal distribution of possible changes in the delivery of organic carbon and nutrients to the system. In addition, the bay system, in terms of actual utilization of such substances, is characterized by various complicating features such as species-specific lags in consumption, changes in microbial utilization and availability of enriched detrital material to the estuarine food webs, and differences in phytoplankton production with time. Consequently, the overall changes in the amounts of such products delivered to the bay can only be calculated in a gross fashion. In general, the loss of the estuarine food webs would be roughly proportional to the reduction of river flow delivered to the bay.

Specific variables such as turbidity and color usually reflect the level and time of Apalachicola River flow so that projected increases or decreases would result in similar changes in such variables in the estuary. Short-term reductions in the river flow could be generally associated with increases in the oxygen anomaly, although dissolved oxygen per se probably would not be affected.

Reductions in river flow would result in proportional decreases in nutrient delivery and phytoplankton productivity. At the same time, organisms that depend on phytoplankton, such as oysters, would be adversely affected by such reductions. The associated decrease of color and turbidity would, all other factors remaining equal, probably allow an expansion of the submerged aquatic vegetation (SAV) which, in turn, would provide habitat and food for those organisms inhabiting the estuary during the warmer months of the year. With a shift to higher salinity, white shrimp, oysters, and various sciaenid fishes such as spot and croaker would be at a disadvantage relative to the pink shrimp, spotted seatrout, and other organisms that are associated with the submerged aquatic vegetation. Because of the complexity of the interrelationships of habitat, physical constraints, productivity, predator-prey relationships, and competition, no exact sequence of response can be predicted. Overall, with reductions of the river flow, the major fisheries that currently are important in the Apalachicola estuary would give way to fisheries associated with the seagrass beds; the Apalachicola system would begin to assume the characteristics of Apalachee Bay, all other factors remaining the same. Increases in river flow would, within limits, expand the current estuarine habitat and increase the level of fisheries production in the Apalachicola Bay system.

A review of the long-term biological responses of the Apalachicola estuary to periods of drought and flood indicate that the response to prolonged increases or decreases in precipitation and river flow would be extremely complex and would, in large part, be based on the the responses of individual populations to such changes. There would also be considerable differences between the short- and long-term responses of such organisms to prolonged changes in state variables. The complex relationship of productivity and salinity in the estuary complicates this situation. Reductions in river flow would result in increases in the numerical abundance and species richness of infaunal macroinvertebrates. However, the relative dominance and overall production of this group of organisms would be reduced by such changes. Increases in river flow would expand the area of the highly opportunistic species that dominate the oligohaline and mesohaline infaunal associations. As river flow decreased over time, there would be a reduction in the physically controlled infaunal groups with a corresponding expansion of the biologically controlled associations. In the long run, such changes would probably lead to an overall reduction in the standing crop biomass and numbers of such organisms.

Increased salinity due to a reduction in river flow would allow oyster predators into the estuary; enhanced disease vectors and parasites could also take a toll of the oyster productivity. Coupled with a reduction in the phytoplankton productivity, the oyster industry would be adversely affected by the projected changes in river flow (hains GFDLx2). The same could be said for those organisms, such as white shrimp, blue crabs, and sciaenid fishes, which currently use the estuary as a nursery area and a haven from the stenohaline gulf predators that would be able to enter the bay as salinities increased. Although the habitat alterations would differ in various parts of the bay, the overall effects of reduced river flow would probably be consistent with general reductions in the various invertebrate and fish species which currently form the basis of the commercial fisheries of the region. The immediate estuarine response to changes in the river flow would probably be somewhat different than the long-term responses to such changes, and such differences would be species-specific in terms of the exact timing of the response. There is some evidence that after an initial increase of certain populations relative to the increased salinity of the system, there would be overall decreases in the numbers and biomass of the dominant epibenthic macroinvertebrates and fishes. Trends of numbers and biomass in the individual populations would not necessarily be parallel in such long-term responses.

TEMPERATURE CHANGES AND EFFECTS

Since the early 1940's, winter low temperatures in the estuarine systems along the north Florida gulf coast have varied considerably from year to year with long-term cycles (8-10 years) of the recurrence of very cold winters. Summer high mean temperatures are less variable from year to year with a range approximating 28-31°C. In terms of changes in the average water temperature of the Apalachicola estuary, the GISS $2xCO_2$ scenario predicted minimal increases of summer high temperatures and increases of 2-3°C of the winter lows. The GFDL $2xCO_2$ model predicted increases of 4-5°C in the average summer high temperatures with projected differences in the summer maximum temperatures of 5-9°C. Average winter low temperatures would increase 4-5°C according to the GFDL projections. This would mean that there will be future summer temperatures (mean) along a range of 31-36°C with summer maximum temperatures occasionally exceeding 40°C.

Temperature is an important factor in the seasonal and interannual variation of organisms in the Apalachicola estuary. Infaunal macroinvertebrates of East Bay have peak numbers/biomass during winter-early spring months of high river flow and low winter temperatures. Species richness/diversity tends to follow similar trends as the numerical abundance. Such indices are associated with winter low temperatures and drought periods. Epifaunal macroinvertebrates, dominated by the commercially important penaeid shrimp and blue crabs, are most abundant during summer-fall periods of high temperature in the estuary. Temperature-related trends of such species appear to be less important than precipitation/river flow variation over long (interannual) periods, although overall invertebrate biomass was associated with periods of very low winter temperatures. Fish numbers tend to peak during winter-early spring periods due to the migrations of juvenile spot and croakers into the estuary at that time. Fish species richness peaks during the fall when a number of species such as the spotted seatrout are abundant: such species are important to the sports fisheries of the region. Absolute fish numbers peaked during the drought period, while fish dry weight biomass was highest during the period of abundant river flow. Thus, the stereotypic seasonal temperature response of the dominant fishes and macroinvertebrates (infaunal and epifaunal) is complicated by long-term trends of regional precipitation; there are distinct interactions of low winter temperatures and drought (e.g., rainfall/river flow) conditions that tend to drive the biological response. There is considerable variation in such response from one level of biological organization to the next.

Based on reviews of upper thermal collection limits (e.g., the highest temperature at which different estuarine species have been taken in the field) and the incipient lethal temperatures (e.g., that temperature causing the death of a specific fraction of test organisms), there is good evidence that as the upper summer mean and maximum temperatures increase, there will be a reduction and/or elimination of some of the most important populations in the Apalachicola estuary. Blue crabs and stone crabs will no longer be in the estuary at mean temperatures of 30-33°C. By the time the mean temperatures reach 35-36°C, oysters, penaeid shrimp, catfishes, mullet, the seatrouts, and southern flounder will no longer be able to use the estuary as a habitat if such populations follow the experimentally determined sensitivities to high temperatures.

Should the GFDL scenario of temperature increase be correct, there is a high probability or likelihood of a major change in the biotic composition and fisheries potential of estuaries across the northern Gulf of Mexico. The main uncertainty in this generalization rests on the possibility that various species will adapt to the temperature changes and/or be replaced by other species that may or may not take their place. Such alternatives are possible and cannot be determined here with any degree of certainty. The relatively short period over which such temperature increases would occur would tend to preclude any significant adaptation of these species to the projected summer high temperatures. Within the context of these uncertainties, it is highly likely that temperature increases in the Apalachicola estuary of 4-5°C (mean high summer values) would have a significant effect on the biota of this system. Since many of the organisms that inhabit the estuary during summer periods are already close to limiting high temperatures, any sustained increase in such temperature levels could have a negative impact on the fisheries potential of the estuary. The projected increases of the winter low temperatures with time could have a profound influence on the species composition of the invertebrate and fish faunas in the estuary with an overall trend toward an increase in tropical species, general increases in species richness and diversity, and associated reductions of fish and invertebrate production. Such projected changes would, of course, be dependent on associated changes in variables such as dissolved oxygen (which would be reduced as the temperatures increased) and salinity, which is dependent on other state variables such as regional precipitation and Apalachicola River flow.

PROJECTED CHANGES IN SEA LEVEL

According to projected changes of sea level from current (baseline) changes to a 2-m increase by the year 2100, there would be progressive and almost total losses of freshwater marshes, the high and low marshes, and the salt marshes of the Apalachicola system. Over half the swamps would be lost with a 2-m increase in sea level by the year 2100. Based on a series of basic assumptions, such losses could be associated with considerable reductions of allochthonous and autochthonous sources of POC to the estuary. Seagrass beds would undergo moderate increases, whereas phytoplankton productivity would be severely reduced. The organic carbon input from the freshwater and saltwater marshes would be virtually wiped out. Overall POC levels, in metric tons/year, would go from 337,385 in 1987 to 51,955 by 2100. Such losses could be directly related to losses of specific populations within the important detrital food webs throughout the Apalachicola Bay system. Overall, the system would be transformed into a seagrass-dominated estuary with relatively minimal input of POC from the surrounding wetlands. The phytoplankton productivity, presently a primary source of organic carbon, would be severely reduced to a relatively insignificant level. The bay system would be much less turbid with most of the nutrients held in the seagrass vegetation.

Ultimately, the various projected changes in the Apalachicola estuary over the next century would include a simultaneous yet gradual alteration of the system, whereby river flow would decrease as the wetlands were inundated by sea level rise. During this time, the water temperature would approach and possible exceed that currently observed in the tropics. Based on the combined effects, it would be virtually impossible to predict the exact sequence of the biological response of the estuary to such changes. However, the estuary would change in terms of species composition, the source and level of productivity, and the fisheries potential. Relative to the current conditions, it would be a warm water seagrass system with less overall fisheries production.

There are certain shortcomings to the analysis presented in this report. A complete review of this problem is given in Appendix VI. The use of correlations, regressions, and other multivariate analyses with long-term, multidisciplinary data is extremely limited in terms of scope, comprehensiveness, and the development of predictive models. In essence, such analysis should be used as a preliminary review of the data so that the state variables can be identified for more comprehensive modeling efforts. In this case, the weaknesses of the rather simple statistical tests were obvious in that specific lags in the biological response to important climatological features could not be included in the final interpretation of results. It should be emphasized that, from the beginning, this analysis was considered preliminary to a more rigorous application of more sophisticated statistical models to the data base.

Accordingly, it is appropriate that a more comprehensive analysis be made of the data with inclusion of similar (currently available) information from other river-estuaries along the gulf coast. The data are sufficient for the application of time-series analysis (ARIMA models, spectral analysis, intervention analysis); such models do not have the rather restrictive assumptions that are associated with most multivariate statistical techniques. In addition, there should be an application of simulation models. Again, the data base is sufficient for the analysis of results with different portions of the available information, both from the standpoint of within-system and between-system comparisons. A comparison of the various estuaries in which long-term field data have been taken would thus be an important part of such an analysis. The use of the above techniques in the development of predictive models could be considered the end result of a 20-year effort to understand river-estuaries along the Florida gulf coast.

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